EXHIBIT 2

Expert Report of
Alexandros Spiliotopoulos, PhD

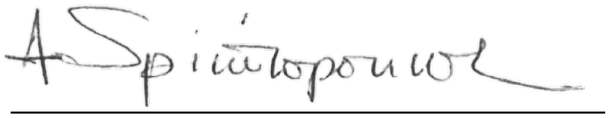
In the United States District Court for the Eastern District of North Carolina

No. 7:23-cv-897

In Re: Camp Lejeune Water Litigation

This document relates to:
ALL PLAINTIFFS

Prepared by:



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December 9, 2024

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Table of Contents

Page

Attachment

List of Figures

List of Tables

List of Acronyms

1,2-tDCE	trans-1,2- Dichloroethylene
ATSDR	Agency for Toxic Substances and Disease Registry
EPA	U.S. Environmental Protection Agency
ft/d	Feet per day
HB	Holcomb Boulevard
g/cm^3	Grams per cubic centimeter
g/ft^3	grams per cubic foot
gpm	gallons per minute
HP	Hadnot Point
HPHB	Hadnot Point and Holcomb Boulevard
LHS	Latin Hypercube Sampling
mL/g	Milliliters per gram
µg/L	micrograms per liter
NRC	National Research Council
PCE	tetrachloroethylene
SSP&A	S.S. Papadopulos & Associates, Inc.
TCE	trichloroethylene
TOC	total organic carbon data
TT	Tarawa Terrace
USMCB	United State Marine Corps Base
UST	underground storage tank
VOC	volatile organic compound
WTP	water treatment plant

REPORT

Section 1
Background and Experience

I, Alexandros Spiliotopoulos of S.S. Papadopulos & Associates, Inc. (SSP&A) was retained by the U.S. Department of Justice to write an expert report and provide my expert opinions in the litigation entitled *In Re: Camp Lejeune Water Litigation*, No. 7:23-CV-897, pending in the Eastern District of North Carolina. I am providing this expert report and my opinions to evaluate the allegations in Plaintiffs' Master Complaint, to respond to the expert reports and opinions of Plaintiffs' experts, Morris Maslia, Mustafa Aral, Norman L. Jones, and R. Jeffery Davis, and to evaluate the Agency for Toxic Substances and Disease Registry's (ATSDR's) water modeling related to Camp Lejeune, which is the subject of Plaintiffs' experts reports and opinions. My opinions are based on my review of the available data and information.

I am a Senior Associate and Senior Hydrogeologist at SSP&A. I hold a Ph.D. degree in civil and environmental engineering from the University of Vermont, and a university degree in civil engineering from the University of Patras, Greece. My expertise includes the application of hydrogeology, modeling, optimization, and parameter estimation to evaluate the origin, distribution, fate, and transport of contaminants in the environment. I have more than 20 years of relevant professional experience evaluating the nature and extent of contamination in aquifers, developing groundwater flow and contaminant transport models, and conducting environmental assessments in the context of regulations and guidance or directives from various government agencies. My Curriculum Vitae is provided as Attachment A. The list of documents that I have considered and/or relied upon will be provided separately.

To conduct my evaluation and render my expert opinions, I relied on my education, research, and professional experience. The documents and information that I considered are of the type that can be reasonably relied upon to support my opinions and are regularly relied upon by practitioners in my field. The materials that were reviewed include, but are not limited to, data, reports, published literature, aerial photographs, correspondence with state agencies, interview summaries, and sworn deposition testimony. I visited the United State Marine Corps Base (USMCB) Camp Lejeune once as part of this evaluation. I was assisted by SSP&A staff.

The hourly rate charged by SSP&A for my services is $268. I have not testified as an expert witness at trial or by deposition in the last 4 years.

Section 2
Opinions

The opinions presented in this report were reached by applying accepted methodology in the fields of hydrogeology, groundwater modeling, and civil and environmental engineering. The opinions expressed in this report are my own and are based on the data and facts available to me at the time of writing. I hold these opinions to a reasonable degree of scientific certainty. I reserve the right to supplement the discussion and findings presented in this report.

1. ATSDR implemented complex calculation methodologies for reconstructing past conditions at Camp Lejeune "to provide [an] epidemiological study with quantitative estimates of monthly contaminant concentrations in finished drinking water because contaminant concentration data and exposure information are limited."[1,2] Due to the absence of sufficient historically observed data and site-specific parameters, the results of these calculations are highly uncertain and cannot be used for determining dose reconstructions at the level of detail that ATSDR presented in their analyses.
   See Section 4

2. ATSDR's dose reconstruction groundwater model for drinking water in Tarawa Terrace was constructed and calibrated using parameters and assumptions that are incorrect or not representative of site conditions, resulting in conservative and biased-high estimated monthly contaminant concentrations.
   See Section 4.1.1; Section 4.1.2;

3. ATSDR's dose reconstruction groundwater model for drinking water in Tarawa Terrace was constructed and calibrated using different parameters and assumptions than for the Hadnot Point model, even though both models simulated similar hydrogeologic conditions. This resulted in faster plume migration and higher contaminant concentrations in the Tarawa Terrace model.
   See Section 4.1.2.2; Section 4.1.2.3; Section 4.2.3.2

4. Application of parameter values based on site-specific data to the ATSDR's dose reconstruction groundwater model for drinking water in Tarawa Terrace would result in substantially lower estimated monthly contaminant concentrations. Furthermore, the model uncertainty range would also be lower.
   See Section 4.1.2.5

5. ATSDR admitted that its dose reconstruction groundwater model for drinking water in Tarawa Terrace resulted in biased-high estimates of monthly contaminant concentrations at one of the water-supply wells. ATSDR used these estimates for their dose reconstruction, resulting in more conservative and biased-high dose reconstruction for the period this well was in service.
   See Section 4.1.2.6

[1] ATSDR-TT, Chapter A, p. A2

[2] ATSDR-HP, Chapter A, p. A13

6. ATSDR's dose reconstruction groundwater model for drinking water in Tarawa Terrace estimated monthly contaminant concentrations that were conservative and biased-high, not reflecting observed data that indicated absence of contamination in the aquifer.
   See Section 4.1.2.7

7. The presentation of the results of the uncertainty analysis conducted by ATSDR for the Tarawa Terrace model was visually misleading by showing a narrow uncertainty range around the calibrated model. Alternate visual presentation of the results indicates estimated monthly concentrations are clearly biased high in the early years.
   See Section 4.1.3.1

8. ATSDR's uncertainty analysis was not bound by historical concentration data, and as a result, focused only on model precision and not accuracy in predicting COC concentrations. ATSDR's uncertainty analysis was presented as though it evaluated the model's accuracy. It did not. Rather, the uncertainty analysis evaluated model precision for parameter ranges that ATSDR selected. ATSDR's uncertainty range is reflective of the narrow range of model parameter values considered in their analysis.
   See Section 4.1.3.2

9. The uncertainty analysis conducted by ATSDR to evaluate the potential range of dose reconstruction estimated monthly contaminant concentrations for drinking water in Tarawa Terrace and did not encompass uncertainty bounds representative of site-specific conditions, resulting in biased-high uncertainty range;
   See Section 4.1.3.2

10. ATSDR applied two different numerical codes for dose reconstruction groundwater modeling for drinking water in Tarawa Terrace, of which the results are not in agreement, due to inconsistent application of contaminant source terms in the two codes. Neither ATSDR, nor Mr. Maslia or Dr. Aral, provided sufficient scientific justification for selecting the higher estimated monthly contaminant concentrations for their dose reconstruction.
    See Section 4.1.4

11. ATSDR's dose reconstruction groundwater model for the volatile organic compound (VOC) degradation by-products in Tarawa Terrace used parameters and assumptions that are incorrect or not representative of site conditions, resulting in conservative and biased-high estimated monthly contaminant concentrations.
    See Section 4.1.4

12. The results of the Tarawa Terrace Flow and Transport Model Post-Audit conducted by Dr. Jones and Mr. Davis indicate that ATSDR's dose reconstruction groundwater model for drinking water in Tarawa Terrace used parameters and assumptions that resulted in conservative and biased-high estimated monthly contaminant concentrations.
    See Section 4.1.5

13. Prior to offering opinions as experts in this litigation, Mr. Maslia and Dr. Aral should have used the data that Dr. Jones and Mr. Davis used to conduct the Tarawa Terrace Flow and Transport Model Post-Audit to update the calibration of the dose reconstruction groundwater model.
    See Section 4.1.5

14. ATSDR's dose reconstruction groundwater model for drinking water in Hadnot Point was constructed and calibrated using parameters and assumptions that are uncertain or incorrect, resulting in conservative and biased-high estimated monthly contaminant concentrations.
    See Section 4.2.1; Section 4.2.2; Section 4.2.3; Section 4.2.4

15. ATSDR incorrectly interpreted field sampling data. For one of the water-supply wells in Hadnot Point, ATSDR included an erroneous concentration value in its model calibration, resulting in conservative and biased-high simulated concentrations, not representative of aquifer conditions.
    See Section 4.2.3.3

16. ATSDR's dose reconstruction model for the VOC degradation by-products was constructed based on the same limited set of observed data, available after December 1984. ATSDR's historical reconstruction prior to December 1984 cannot be verified.
    See Section 4.2.4

17. ATSDR's sensitivity analysis for the various contaminant sources in Hadnot Point indicated that the timing of source-release start date is uncertain and, therefore, it is impossible to determine the historical period that contamination was present in groundwater.
    See Section 4.2.5.1.1

18. The sensitivity analysis of the dose reconstruction groundwater model conducted for drinking water in Hadnot Point was based on parameter variability unsupported by data. Particular combinations of extreme parameter values resulted in conservative and biased-high estimated monthly contaminant concentrations. The results of the sensitivity analysis were incorrectly presented as an uncertainty analysis range.
    See Section 4.2.5.1.2

19. The analysis conducted to evaluate the potential range of dose reconstruction estimated monthly contaminant concentrations for drinking water in Hadnot Point only partially addressed model uncertainty, and it indicated that calibrated reconstructed concentrations were conservative and biased high.
    See Section 4.2.5.2

Section 3
Introduction

This section provides overviews of: important concepts in groundwater modeling; historical operations and groundwater contamination at Camp Lejeune; and studies conducted in Camp Lejeune and related scientific discourse.

3.1 Overview of Groundwater Modeling

Models are "*simplified representation[s] of the complex natural world*."[3] Using mathematics and computer software, modelers can simulate and quantitatively assess environmental processes.[4] However, models can never reflect the complexity and uniqueness of the systems they are intended to replicate and are therefore of limited use.[5] It is important to understand the limitations of models.

3.1.1 Model Uses

Water models are used in a number of ways. Models can be used to forecast future impacts of an action, like the change in groundwater levels caused by pumping from a well.[6] Modeling may also be used to replicate past conditions. This is sometimes called hindcasting.[7] Anderson et al. (2015) indicate that "*[h]indcasting applications are 'uniquely challenging' because it is not possible to collect additional observations to augment the existing historical dataset, which is often meager*."[8]

3.1.2 Model Development

Model development begins with a "conceptual" model. A conceptual model incorporates our understanding of the field setting to construct a description of the groundwater flow system.[9] This is done using collected field data and related information from previous investigations and studies in the area.[10] The conceptual model is, therefore, a qualitative summary of what is known about the processes occurring in the hydrogeological system, such as the boundaries, aquifer properties, groundwater flow, etc.[11]

As described in the expert report of Dr. Remy Hennet, the hydrogeological system encompasses the composition of the geologic materials (e.g. sand grains, clay particles, rock fragments) in the subsurface with the presence of water. This composition of geologic materials is a porous medium.[12]

An aquifer is a saturated porous medium that can transmit water flowing from points of high pressure to points of low pressure.[12] This flow occurs through the interconnected pores of the porous medium, within hydrostratigraphic zones of different geologic material and properties. Aquifers are

[3] Anderson et al. (2015), p.5; NRC (1990), p. 52

[4] Harter et al. (2018), p. 47

[5] Anderson et al. (2015), p. 12

[6] Anderson et al. (2015), p. 9

[7] Anderson et al. (2015), p. 9

[8] Anderson et al. (2015), p. 11 (citing Clement (2011) p. 620)

[9] Anderson et al. (2015), p. 17

[10] Anderson (2015), p. 17

[11] Anderson et al. (2015), p. 35

[12] Freeze and Cherry (1997), p.17

encountered at different depths in the subsurface. The soils between the ground surface and the aquifer are what is called the "unsaturated" zone.[13]

Once the conceptual model is constructed and the purpose of the model is defined, the mathematical model that describes the processes incorporated in the conceptual model is selected.[14] These processes are complex. A mathematical model is a set of "governing" equations that calculates the progression of these complex processes in space and time. Definitions of "boundary" and "initial" conditions are required for the solution of these mathematical models.

Numerical codes are algorithms that carry out the calculations of the mathematical model. Scientific software codes have been developed to perform such computations. For example, MODFLOW, a code created by the U.S. Geological Survey, is used to quantitatively analyze groundwater flow through the porous medium.[15]

A groundwater model is the "translation" of the conceptual model of the groundwater system to a numerical model. This translation requires "*designing the grid/mesh, setting boundaries, assigning values of aquifer parameters, and hydrologic stresses, and, for transient models, setting initial conditions*."[16]

Construction of the numerical model involves creating the three-dimensional "grid" which serves as the framework of the numerical model.[17] This "grid" (or, sometimes, mesh) consists of cells (most commonly cube-shaped) intended to represent the porous medium in a piece-wise manner.[18] The center-point of the cell is known as the "node." Figure 1 illustrates an example of a MODFLOW grid, showing aquifer hydrostratigraphy (i.e. model layers).

[13] Freeze and Cherry (1979), p.15

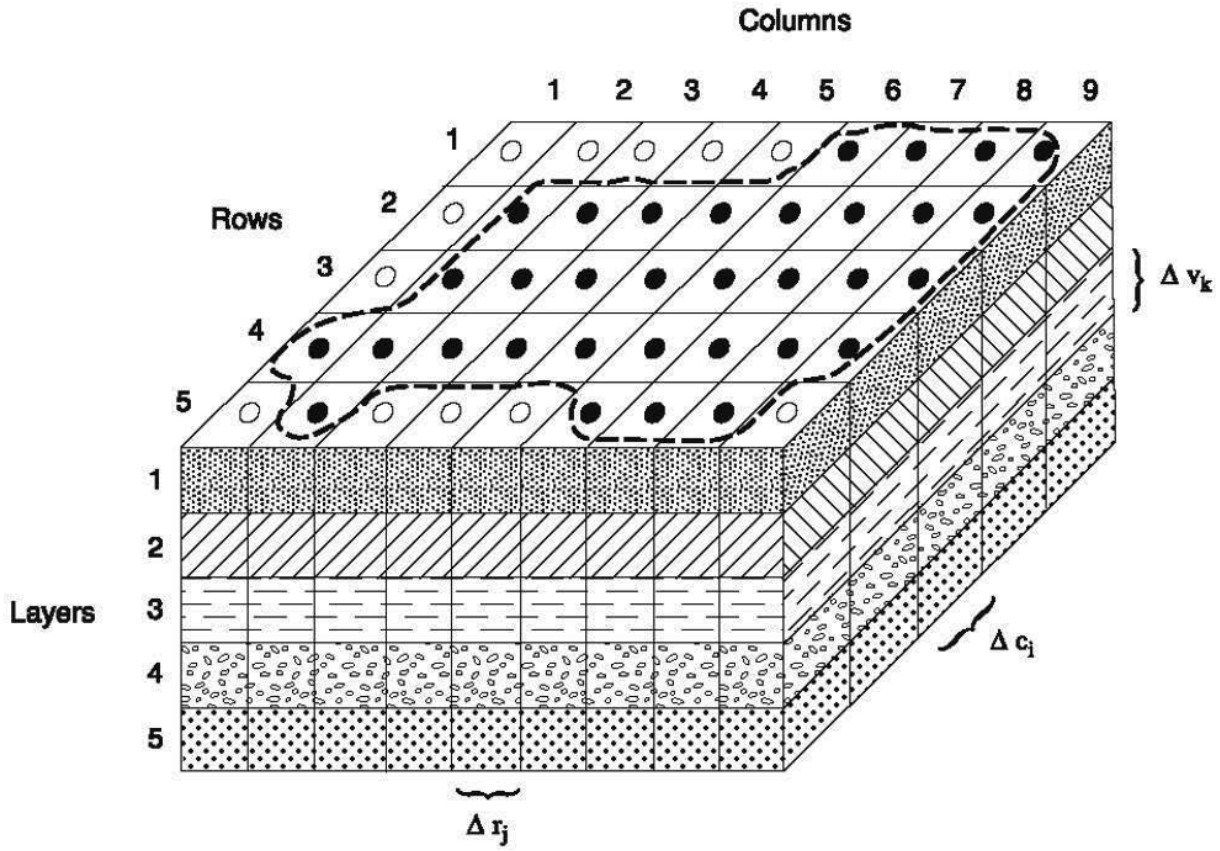
[14] Anderson et al. (2015), p. 17

[15] Anderson et al. (2015), p. 9

[16] Anderson et al. (2015), p. 17

[17] Anderson et al. (2015), p. 17

[18] Anderson et al. (2015), p. 71



Source: https://perma.cc/Z3FR-XZDA; https://www.usgs.gov/media/images/modflow-2005-hypothetical-aquifer-system

Figure 1: Example of a MODFLOW Grid

Parameters are defined in numerical models with their values assigned in each model cell. Parameters are "*constant term[s] in an equation that reflect[] a relationship*."[19] For example, in the equation $y = 5x$, five is a coefficient which can be divided out to solve for the variable x. The coefficient, *5*, has a constant value that reflects the relationship between x and y. Parameters are used to reflect relationships and conditions in the natural world, such as the rate at which water can move through certain types of soils. For example, in sands and gravels water movement is much faster than in rocks or clays. In groundwater modeling, the "hydraulic conductivity" is a constant of proportionality, describing the relationship between the rate of groundwater flow and the pressure differential that causes it.[20]

In addition to assigning each cell or node its own parameter values,[21] the modeler also sets boundaries and various other conditions.[22] This allows modelers to re-create features of the natural environment, such as confining layers in the sub-surface, or stresses to the system, such as the operation of pumping wells.

[19] National Judicial Conference (2010), p. 40

[20] Freeze and Cherry (1979), p. 16

[21] Anderson et al. (2015), p. 203

[22] Anderson et al. (2015), p. 17

3.1.3 Model Calibration

Following its development, the model is then calibrated. Calibration is a process by which the modeler conducts "history matching," by adjusting model parameters until the model outputs reasonably match measured field observations.[23] Modelers can use a parameter estimation code (such as PEST) to assist in determining calibration targets.[24] The goal of calibration is to adjust the model parameters so that the model results are as close to observed field data as possible.[25] This is important because the calibrated model should be as accurate as possible to be an appropriate tool for estimating the quantities identified by its intended use.[26]

3.1.4 Sensitivity Analysis

Sensitivity analysis is the process by which one or more parameters are manipulated, to see how model outputs change in response to parameter changes.[27] If modifying a parameter causes a relatively large change in model results, this indicates to the modeler that he or she needs more data to constrain that parameter.[28]

3.1.5 Uncertainty Analysis

Models are inherently uncertain.[29] For a variety of reasons, "*a groundwater model cannot give a single true answer*."[30] One reason for that is non-uniqueness. This means that different variations of parameters can provide results that are close to field observations.[31] Moreover, there are assumptions required in designing models and approximating the environment that cannot be fully understood.[32] Uncertainty analysis may provide a range of possible outcomes and help assess a model's error margins.[33] Uncertainty analysis is a statistical analysis that provides a range of probabilities, which are used to characterize confidence in the model's outputs.[34]

The calibrated model output (prediction) should conform with the following general rule: "*ideally, the value of that prediction should lie somewhere near the centre of the uncertainty band of the prediction. In this way, the potential for predictive error is minimized*."[35]

Confidence in model outputs is essential when important decisions rely on the outputs of these models. Important aspects of a model, with direct implications for the issues discussed in this report, are its precision and accuracy.[36] Hill and Tiedeman use an analog from archery to describe these terms:

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[23] Anderson et al. (2015), p. 19, 202; Harter et al. (2018), p. 57; ASTM D5981/D5981M-18 (2018); Reilly and Harbaugh (2004), p. 23

[24] Anderson et al. (2015), p. 18.

[25] Anderson et al. (2015), p. 18

[26] Reilly and Harbaugh (2004), p. 4

[27] Reilly and Harbaugh (2004), p. 3; ASTM D5611-94 (2016)

[28] Harter et al. (2018), p. 58; ASTM D5611-94 (2016); Reilly and Harbaugh (2004), p. 2

[29] Anderson et al. (2015), p. 12; National Research Council (1990), p. 216

[30] Anderson et al. (2015), p. 12

[31] Anderson et al. (2015), p. 12

[32] National Research Council (1990), p. 221–30

[33] Anderson et al. (2015), p. 18

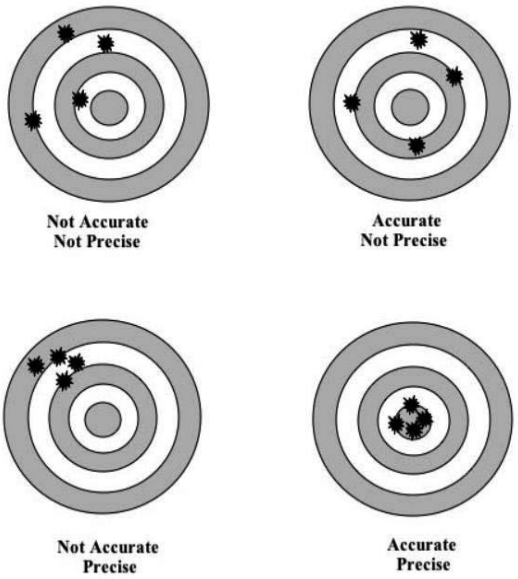
[34] Anderson et al. (2015), p. 457

[35] Doherty (2015), p. 52

[36] Hill and Tiedeman (2007), p.13

- Precision: "*a set of shots is precise if the shots fall within a narrow range, regardless of whether they are near the bull's eye*."[37] The equivalent to this analog, when considering a groundwater model that attempts to reconstruct historical conditions, is its uncertainty analysis. The model can be run many times, and model results fall within a narrow range. However, model results can be far off from the observed data.
- Accuracy: "*a set of shots is accurate if the shots are distributed in a narrow range around the bull's eye*."[38] For the groundwater reconstruction model, this means that its results are close to the observed data.

The National Oceanic and Atmospheric Administration (NOAA) provided a schematic to illustrate the concepts of 'precision' and 'accuracy' as they are understood in natural sciences. This schematic is depicted in Figure 2.



Source: https://perma.cc/J794-KUYM; https://celebrating200years.noaa.gov/magazine/tct/tct_side1.html[39]

Figure 2: Accuracy versus Precision

Based on the above, a precise model is not necessarily accurate. In order for a model to meet the objectives of its intended use, it must also be accurate. The definition of accuracy refers directly to the "bull's eye," i.e. the real world in the case of a groundwater model. The real world is described by observed data. In the absence of data, the ability to determine whether a model is accurate is diminished. Hence, history matching, i.e. fitting model results to observed data is critical.

3.1.6 Model Updates

Developing hydrologic models is an iterative process.[40] Model calibration, sensitivity analysis and uncertainty analysis are related processes that are all necessary for constructing a reliable model. Modelers

[37] Hill and Tiedeman (2007), p.13

[38] Hill and Tiedeman (2007), p.13

[39] On this website, a discussion is provided regarding geological surveying, but the concepts and definitions apply equally to groundwater modeling.

[40] National Research Council (1990), p. 191

should continually make corrections to models and repeat the steps outlined above throughout the development of the model. Moreover, models should be *"routinely updated as new data become available*."[41]

A post-audit is a "*comparison between conditions simulated in a forecast and conditions that actually occurred*."[42] It may occur years after the model was developed, and uses data collected past the original model simulation timeframe.[43] Post-audits may lead to updates in model calibration using these new data, to improve model performance.[44]

3.1.7 Simulating Contaminant Transport

Using the process described above, modelers may attempt to forecast or hindcast the movement of contaminants in groundwater. To track the flow of contaminants in an aquifer, modelers may link ("couple") groundwater flow models, such as MODFLOW, with contaminant fate and transport models.[45] MT3DMS is an example of a contaminant fate and transport model that may be coupled with MODFLOW.

Contaminant sources may be included in the model to simulate their historical contribution to contamination in the aquifer. Mass loading is a parameter that describes how much of a contaminant is introduced in the subsurface and enters the aquifer.[46]Mass loading is an important parameter for understanding how much contamination is assumed to be entering the system and migrating in the aquifer.Another important consideration in any groundwater model is the presence of wells. Wells pump water out of the aquifer, decreasing pressure around the well. The decrease in pressure creates a "cone of depression," which forces groundwater towards the well screen. Therefore, contamination reaches the pumping well, transported by groundwater in the aquifer. However, contamination doesn't always move at the same speed as groundwater. It is sometimes "retarded" due to soil and contaminant characteristics.[47]

3.1.8 Concluding Remarks

Anderson et. al (2015) emphasized the importance of history matching: "*[h]istory matching is important for evaluating a model's fit for purpose: if a model cannot reproduce the measured heads and fluxes with sufficient accuracy, one can have little confidence that the calibrated model will adequately reproduce unmeasured heads and fluxes.*"[48] Although this discussion referred to the results of a groundwater flow model, it is applicable to any model, including those hindcasting contaminant transport in aquifers. In all cases, a model is required to reasonably fit the measured data to reliably tell us what happens when data are not available. As Doherty (2015) states: "[a] *hypothesis is proposed, and evidence is collected to test it.*"[49] The model is a hypothesis, which is tested against observed data. During calibration, the hypothesis is updated until the model fits the observed data. Model calibration is not possible when there are no historical data to match.

As mentioned earlier, a calibrated model is still uncertain. Furthermore, model calibration is further hindered when there are limited or, worse, no historical data to match. In such cases, the uncertainty of the resulting model is vastly exacerbated. Any uncertainty analysis conducted on such a "calibrated" model

[41] Anderson (2015), p.18

[42] Anderson (2015), p. 481.

[43] Anderson (2015), p.18

[44] Anderson (2015), p. 481–82.

[45] Zheng & Bennet (2002), p. 195

[46] National Research Council (1990), p. 140

[47] Anderson et al. (2015), p. 363

[48] Anderson et al. (2015), p. 376

[49] Doherty (2015), p. 179

can only address its precision, but not its accuracy, as Dr. Dan Waddill, a NAVFAC engineer who reviewed and provided feedback to ATSDR on their modeling, aptly indicated in his deposition.[50]

Doherty and Moore (2021) emphasized the importance of historical data: *"[i]t must not be forgotten, however, that it is information, and information alone, that can reduce predictive uncertainty. It follows that if a complex model structure can express information that emerges from site characterisation studies, then it does indeed have the potential to reduce the uncertainties of at least some decision-critical predictions*."[51] In other words, a complex model that attempts to simulate important processes in the subsurface, will be highly uncertain if it is not tied to measured data, i.e., information about the simulated processes.

3.2 Historical Operations and Groundwater Contamination at Camp Lejeune

USMCB Camp Lejeune is a military base located near Jacksonville, in Onslow County, North Carolina. Operations began at USMCB Camp Lejeune during late 1941.[52]

The Tarawa Terrace (TT), Hadnot Point (HP), and Holcomb Boulevard (HB) water distribution systems are of interest because of historical contamination issues. Two of these three water distribution systems were contaminated with VOCs due to contamination in groundwater. Groundwater extracted from water-supply wells was directed to water treatment plants (WTPs) in these areas. Groundwater within the Tarawa Terrace service area was contaminated mostly with tetrachloroethylene (PCE) and PCE biodegradation by-products.[52] Groundwater within the HP service area was contaminated with trichloroethylene (TCE), and to a lesser degree trans-1,2-DCE (1,-tDCE), PCE, and refined petroleum products, such as benzene, toluene, ethylbenzene, and xylenes (BTEX).[52] The HB WTP service area was intermittently supplied with contaminated water from the HP WTP between 1972 and 1985, during interconnection events between the two systems.

ATSDR constructed numerical models to simulate groundwater flow and contaminant transport in the aquifers under Camp Lejeune, and to reconstruct the historical concentrations of contaminants in finished water produced by the treatment plants and delivered to the water-distribution networks. ATSDR constructed two separate models, one of the Tarawa Terrace (TT) family housing area, and another of the Hadnot Point and Holcomb Boulevard (HPHB) areas. ATSDR did not model groundwater flow and contaminant transport in any other areas on base.

ATSDR constructed a water distribution model for HB, to calculate concentrations in the piping network connecting the WTP with the consumers. This was done to calculate the effects of the intermittent connection of HB with the HP WTP mentioned above. For Tarawa Terrace and Hadnot Point, ATSDR did not simulate transient contaminant transport in their water distribution networks. For those WTPs, ATSDR ran a simple mixing model for calculating the weighted contaminant concentration in the influent to the WTP, considering monthly flow rates for each well and corresponding model-simulated concentrations.[53] For each month, the volume of water pumped in a well and the simulated PCE concentration in that volume

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[50] Deposition of Dan Waddill, 2024, p. 133:18-134:3 ("*What ATSDR did in this case is uncertainty with respect to model precision. That's how tightly the model runs compared to one another, and they did. They created ranges and all of that. But it's precision, and it has nothing to do with uncertainty with respect to model accuracy, which is how -- how closely does that cluster come to the real world, and that's just unknown because they didn't have data to do that kind of analysis.*").

[51] Doherty and Moore (2021), p. 33

[52] ATSDR-HP, Chapter A, p. A7

[53] ATSDR stated that monthly water-supply well flow rates and corresponding simulated concentrations were provided in Chapter K of the TT Report. A reference to Chapter K was provided in Chapter A, indicated as "In press 2007." However, Chapter K was never published (ATSDR-TT, Chapter A, p. A74 and A80)

of water were calculated, and the weighted PCE concentration reflected the aggregate of these quantities for all wells.

To construct these models, ATSDR had to define contaminant source locations and quantify their contributions of contamination to the aquifers. ABC One-Hour Cleaners, "*an off-base dry-cleaning facility that used PCE in the dry-cleaning process*"[54] was indicated as the source of the contamination found in the Tarawa Terrace water-supply wells. However, the timing and quantification of contaminant releases from that source are uncertain, due to lack of historical data. Figure 3 shows the TT WTP Area, including water-supply well locations and the identified contaminant source.

[54] ATSDR-TT, Chapter A, p. A10

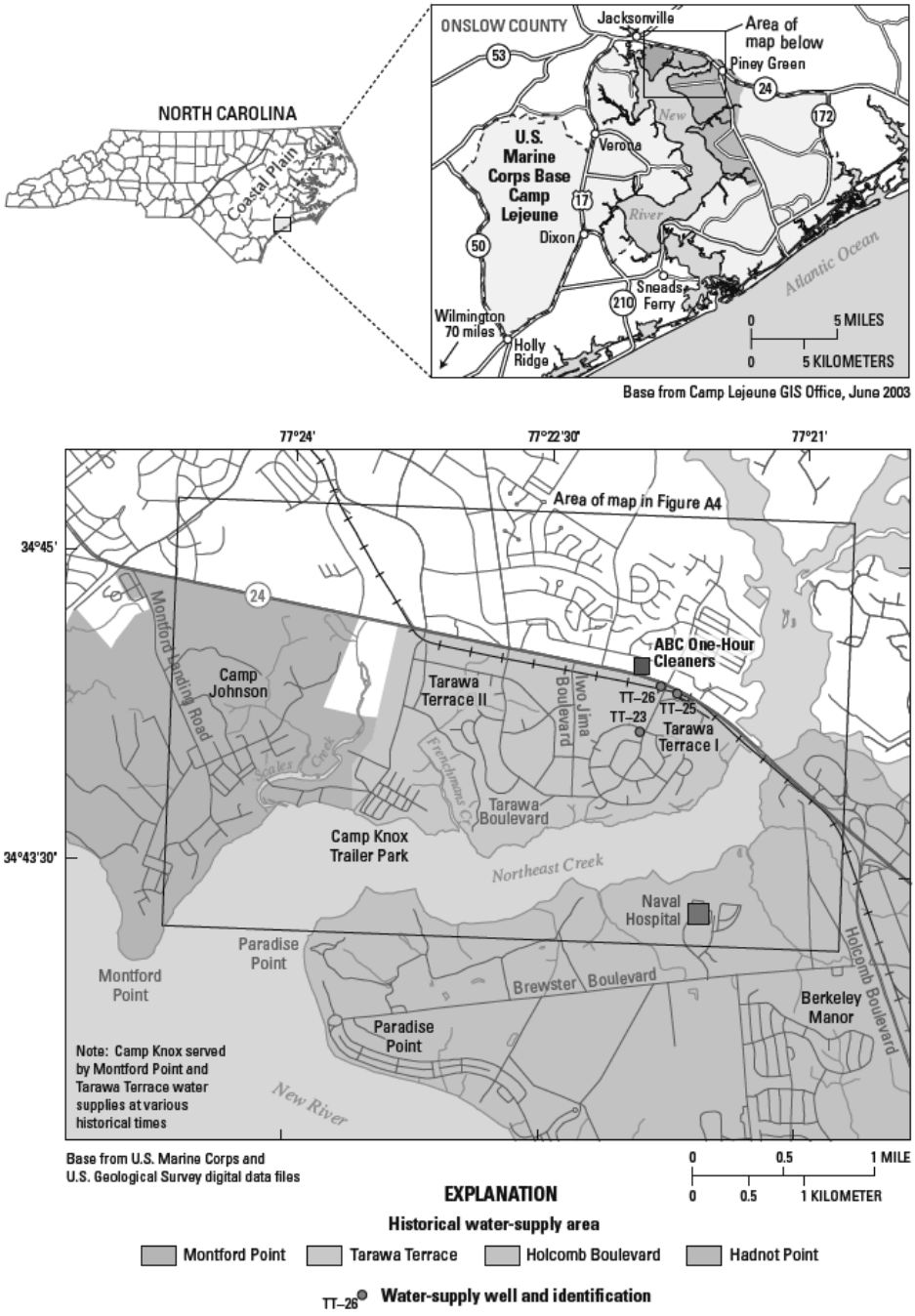

Figure A1. Selected base housing and historical water-supply areas, Tarawa Terrace and vicinity, U.S. Marine Corps Base Camp Lejeune, North Carolina.

Figure 3: Map of Tarawa Terrace Area, Camp Lejeune

Historical base operations and waste-disposal practices have been identified as being responsible for contamination of groundwater and finished-water supplies within the HPHB areas. At Hadnot Point, different sources were defined, based on soil and groundwater sampling data, as well as historical data on infrastructure and operations. Leaking underground storage tanks (USTs), surface spills, and landfill material were some of the contaminant sources identified by ATSDR. Similar to Tarawa Terrace, the timing and quantification of contaminant releases from those sources are uncertain due to a lack of historical data.

Figure 4 shows the HPHB WTP Area, including water-supply well locations and locations of storage tanks. It should be noted that not all of the storage-tank locations were identified as potential sources of contamination.

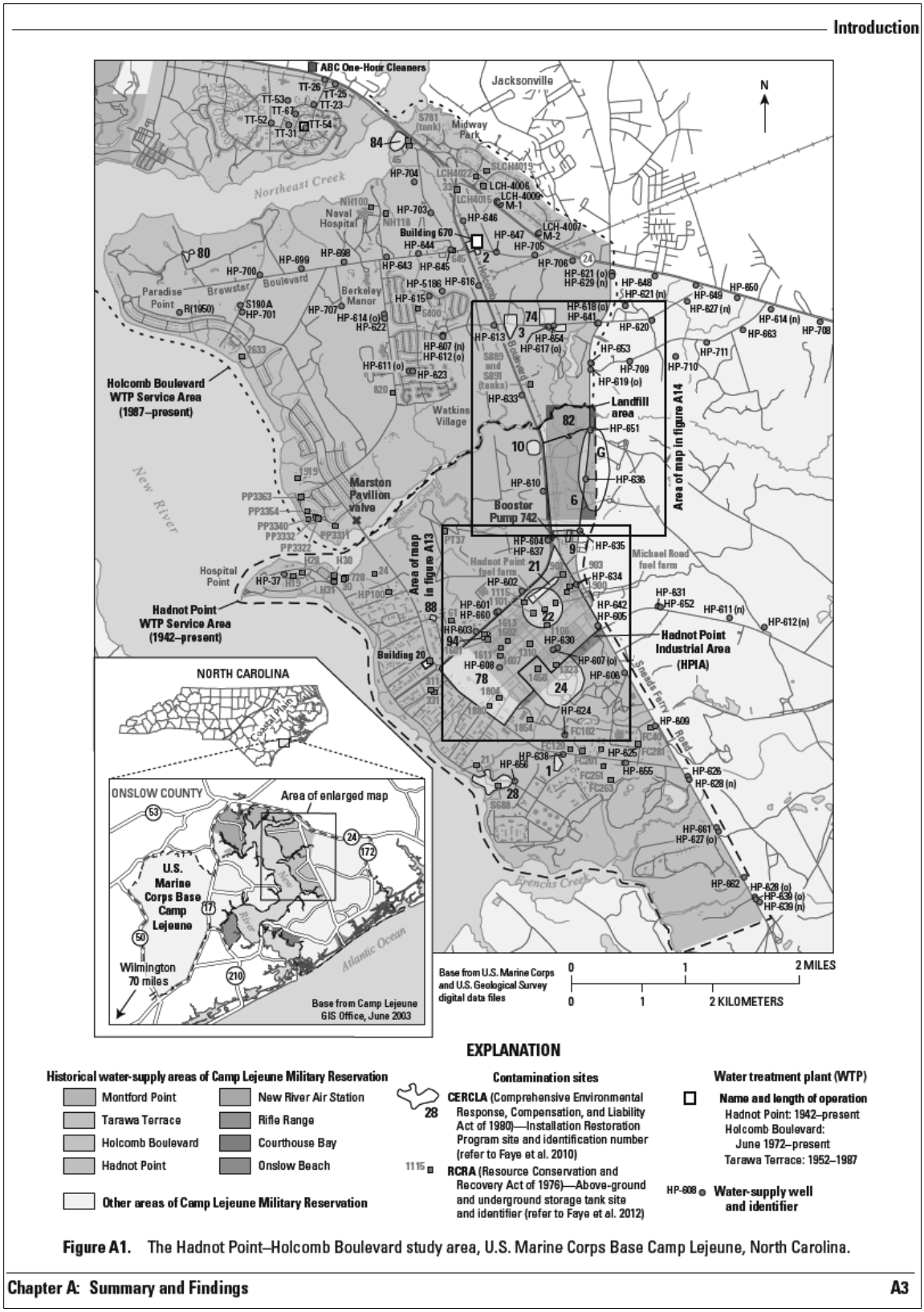

Figure 4: Map of Hadnot Point-Holcomb Boulevard Area, Camp Lejeune

ATSDR constructed these groundwater models based on very limited historical data. See the expert report of Dr. Hennet for a detailed discussion of the historical data available to ATSDR. The Tarawa Terrace housing area was constructed in 1951,[55] with pumping of water supply wells beginning in 1952, although more precise dates are not available. Wells were operated in groups at different times, but associated operational data for each well were not available.

At Hadnot Point, operations began in late 1941. Construction of family housing and major infrastructure began in 1942,[56] including the water distribution network. The water treatment plant was constructed during 1941 and 1942.[57] In 1942, 21 water supply wells began operation, with additional wells installed in following years, including replacements of several wells.[58] Similar to Tarawa Terrace, wells operated in groups, but very limited operational data are available.[59]

In order to simulate the operation of water supply wells and their impact on groundwater flow and the migration of contamination in the aquifer, ATSDR developed hypotheses for the historical pumping schedules and corresponding flow rates for the water supply wells. These pumping schedules were developed using complex calculation procedures that were based on limited to no historical data, or other ancillary information.

Limited data were available with respect to the hydrogeologic setting within the local scale of the aquifer. ATSDR constructed groundwater flow models based on this limited dataset, implementing generalized assumptions on the geologic framework, the distribution of the hydraulic properties within the model domain, and boundary conditions.

Finally, limited data were available for constructing the geochemical conceptual model underpinning the contaminant transport model used for simulating contaminant plume migration in the aquifer. Critical parameters impacting the migration of contaminants in groundwater were not based on site-specific data. Rather, they were ultimately defined during model calibration.

ATSDR asserted that their modeling approach provided a high level of detail and accuracy to estimate monthly exposure concentrations in finished water.[60,61] However, assumptions and/or parameter values used by ATSDR in constructing these models were incorrect or inconsistent with site-specific data. As a result, reconstructions of estimated historical monthly contaminant concentrations in finished water produced by the treatment plants and delivered to the water-distribution networks were conservative and biased high, as discussed in detail in Section 4.

3.3 Timeline and Scientific Discourse on ATSDR's Camp Lejeune Water Modeling

What follows is a brief timeline summarizing ATSDR's study of water contamination at Camp Lejeune and related events.

- **1989:** The Environmental Protection Agency (EPA) placed USMCB Camp Lejeune and an off-base dry cleaner, ABC One-Hour Cleaners, on its National Priorities List.[62]

[55] ATSDR-TT, Chapter A, p. A10

[56] ATSDR-HP, Chapter A, p. A2

[57] ATSDR-HP, Chapter A, p. A11

[58] ATSDR-HP, Chapter A, p. A11

[59] ATSDR-HP, Chapter A, p. A13

[60] ATSDR-TT, Chapter A, p. A97

[61] ATSDR-HP, Chapter A, p. A181

[62] Maslia Deposition (2024), 86:3-23; ATSDR-TT, Chapter A; ATSDR-HP, Chapter A

- **1997:** ATSDR completed a Public Health Assessment (PHA) for Camp Lejeune.
  - The 1997 PHA noted that ATSDR "was established by Congress in 1980 under [CERCLA]" and that "ATSDR has been required by law to conduct a public health assessment at each of the sites on the EPA National Priorities List."[63]
  - The 1997 PHA recommended conducting an epidemiological study on specific birth defects and cancer in children who were exposed in utero to the COCs at Camp Lejeune.[64]
  - ATSDR made the decision to utilize water modeling to quantify exposure assessments for the epidemiological study based on prior modeling efforts related to Dover Township, New Jersey.[65]

- **2002/2003:** ATSDR started working on water modeling efforts related to Camp Lejeune.[66]
  - Morris Maslia, Project Officer for ATSDR's Exposure-Dose Reconstruction Program, was Project Lead for water modeling on Camp Lejeune.
  - Mustafa Aral, a Georgia Tech Professor, was also on ATSDR's water modeling team for Camp Lejeune.

- **March 28-29, 2005:** ATSDR held an "Expert Peer Review Panel" regarding "ATSDR's Water-Modeling Activities in Support of the Current Study of Childhood Birth Defects and Cancer at U.S. Marine Corps Base Camp Lejeune, North Carolina."[67]

- **May 2007:** The United States Government Accountability Office (GAO) issued a "Report to Congressional Committees" entitled "Activities Related to Past Drinking Water Contamination at Marine Corps Base Camp Lejeune."
  - In reference to the 2005 Expert Peer Review Panel, GAO commented: "But all of the panel experts raised concerns about the limited historical record of the amount of PCE or TCE concentration identified at individual Camp Lejeune wells. They said that with limited historical data there would be minimal potential for water modeling to provide accurate information about the level of concentration of the contamination and thus about each individual's total amount of exposure. As an alternative to estimating the extent of each study individual's exposure using the water modeling results, four panel experts suggested ATSDR could use simpler categories of whether and to what extent individuals were exposed to water contamination."[68]

- **June 2007–February 2008:** ATSDR completed and published water modeling reports related to Tarawa Terrace entitled "Analyses of Groundwater Flow, Contaminant Fate and Transport, and

[63] ATSDR PHA (1997); Maslia Deposition (2024), 86:24-87:17

[64] ATSDR PHA (1997); Maslia Deposition (2024), 87:23-88:23

[65] Maslia (2024), Deposition, September 26, 91:2-16

[66] Maslia (2024), Deposition, September 26

[67] ATSDR Expert Review Panel Report and Transcripts (2005)

[68] GAO (2007), p. 55

Distribution of Drinking Water at Tarawa Terrace and Vicinity, U.S. Marine Corps Base Camp Lejeune, North Carolina: Historical Reconstruction and Present-Day Conditions," Chapters A- F, H.[69]

ATSDR's Tarawa Terrace reports indicated that the water modeling was intended to support an epidemiological study, not for the purpose of making exposure assessments in individuals.

- "The Agency for Toxic Substances and Disease Registry (ATSDR), an agency of the U.S. Department of Health and Human Services, is conducting an epidemiological study to evaluate whether in utero and infant (up to 1 year of age) exposures to drinking water contaminated with volatile organic compounds (VOCs) at U.S. Marine Corps Base Camp Lejeune, North Carolina (Plate 1), were associated with specific birth defects and childhood cancers. The study includes births occurring during the period 1968–1985 to women who resided in family housing at Camp Lejeune."[70]
- "ATSDR is using water-modeling techniques to provide the epidemiological study with quantitative estimates of monthly contaminant concentrations in finished drinking water[] because contaminant concentration data and exposure information are limited. Results obtained by using water-modeling techniques, along with information from the mother on her water use, can be used by the epidemiological study to estimate the level and duration of exposures to the mother during her pregnancy and to the infant (up to 1 year of age). Using water-modeling techniques in such a process is referred to as historical reconstruction (Maslia et al. 2001)."[71]
- "ATSDR's exposure assessment cannot be used to determine whether you, or your family, suffered any health effects as a result of past exposure to PCE-contaminated drinking water at Camp Lejeune."[72]

- **March 26, 2008:** ATSDR held a Technical Meeting with the United States Marine Corps and Navy to present its water modeling efforts related to Tarawa Terrace.[73]

- **June 19, 2008:** The United States Marine Corps and United States Department of the Navy (DON) provided comments, which were drafted by the Navy's water modeling expert Dr. Dan Waddill, to ATSDR about its water modeling efforts related to Tarawa Terrace.

  The Marine Corps' and Navy's comments included the following:

  - "Model simulations provide monthly concentrations from 1952 to 1987, but measured concentrations for model calibration are available only in 1982 and 1985. Thus, the majority of the simulated concentrations cannot be compared to measured data."[74]

[69] Chapters G and I were completed and published in April 2008 and February 2009, respectively.

[70] ATSDR-TT, Chapter A, p. iii

[71] ATSDR-TT, Chapter A, p. A2

[72] ATSDR-TT, Chapter A, p. A98

[73] GAO (2007)

[74] Department of Navy (2008), CLJA_2019ATSDR04-0000002372–2379

- "Simulated concentrations did not fall within calibration targets for a majority of the measured PCE concentrations at the water supply wells, suggesting that the 'accuracy' of the model is less than the chosen calibration standard of ± 1/2-order of magnitude."[75]
- "Due to lack of measured PCE concentrations, the Tarawa Terrace model was not validated. Therefore, the model was not 'put at risk,' and it is difficult to judge the accuracy of the simulated PCE concentrations beyond the limited times when calibration data are available."[76]

The Marine Corps and Navy made the following recommendations:

- "Improve communication with the public and other stakeholders by developing a method for presenting the uncertainty in the model-derived PCE concentrations. The method should be clear and readily understood, perhaps using error bars or presenting a concentration range rather than a single number. The method should be applied consistently whenever concentrations are discussed or presented in model reports, websites, public meetings, etc."[77]
- "Convene an expert panel to examine the model results and determine the best use for the data. Overall, the panel should develop a path forward that is scientifically sound and will best meet the critical concerns of the public."[78]
- "Apply all lessons learned from the Tarawa Terrace modeling efforts to the scoping of the approach for Hadnot Point."[79]

Around this timeframe, ATSDR took down a public webpage that generated estimated monthly contaminant concentrations based on an individual's address. Mr. Maslia testified that "in working with the Department of Navy, they expressed some reservations that there were insufficient qualifiers on the data, not the table itself. But when somebody just put in an address and got a value out, it did not explain to them the limits of the data or the simulated data."[80]

- **March 2009:** ATSDR published its "Response to the Department of the Navy's Letter on Assessment of ATSDR Water Modeling for Tarawa Terrace."[81]

  In responding to the Marine Corps' and Navy's comments, ATSDR reaffirmed that the Tarawa Terrace water modeling was intended to support an epidemiological study and not for the purpose of making exposure assessments in individuals.

  - "ATSDR maintains that the models (flow, transport, and mixing) are sufficiently calibrated, given the quantity and accuracy of data provided and the intended use of the simulated historically reconstructed concentrations. Although the DON is correct in pointing out that some simulated results did not meet the calibration target, ATSDR believes that the DON should assess these results in terms of: (1) similar peer-reviewed reports, (2) currently established model calibration practices, and (3) the intended use of

[75] Department of Navy (2008), CLJA_2019ATSDR04-0000002372–2379

[76] Department of Navy (2008), CLJA_2019ATSDR04-0000002372–2379

[77] Department of Navy (2008), CLJA_2019ATSDR04-0000002372–2379

[78] Department of Navy (2008), CLJA_2019ATSDR04-0000002372–2379

[79] Department of Navy (2008), CLJA_2019ATSDR04-0000002372–2379

[80] Maslia (2010), Deposition, June 30, 79:25-80:5.

[81] ATSDR (2009)

the modeling results by the epidemiological study. That is, are the ATSDR analyses within the accepted norm of current-day modeling practices, are the ATSDR analyses an exception to this norm, and will there be sufficient reliability for an epidemiological study?"[82]

- "To address the issue of the intended use of the water-modeling results by the current ATSDR epidemiological study, the DON should be advised that a successful epidemiological study places little emphasis on the actual (absolute) estimate of concentration and, rather, emphasizes the relative level of exposure. That is, exposed individuals are, in effect, ranked by exposure level and maintain their rank order of exposure level regardless of how far off the estimated concentration is to the "true" (measured) PCE concentration. This rank order of exposure level is preserved regardless of whether the mean or the upper or lower 95% of simulated levels are used to estimate the monthly average contaminant levels. It is **not** the goal of the ATSDR health study to infer which health effects occur at specific PCE concentrations—this is a task for risk assessment utilizing approaches such as meta-analysis to summarize evidence from several epidemiological studies because a single epidemiological study is generally insufficient to make this determination. The goal of the ATSDR epidemiological analysis is to evaluate exposure-response relationships to determine whether the risk for a specific disease increases as the level of the contaminant (either as a categorical variable or continuous variable) increases."[83]

- **April 29-30, 2009:** ATSDR held an "Expert Panel" regarding "ATSDR's Methods and Analyses for Historical Reconstruction of Groundwater Resources and Distribution of Drinking Water at Hadnot Point, Holcomb Boulevard, and Vicinity, U.S. Marine Corps Base Camp Lejeune, North Carolina."

- **2009:** The National Research Council (NRC) published a report entitled *Contaminated Water Supplies at Camp Lejeune: Assessing Potential Health Effects*.[84]

  The 2009 NRC Report reviewed and provided comments on ATSDR's water modeling related to Tarawa Terrace, including the following:

  - "The National Research Council (NRC) conducted this review in response to a request from the U.S. Navy, the department under which the Marine Corps operates. The Navy was mandated by the U.S. Congress (Public Law 109-364, Section 38) to request a review

[82] ATSDR (2009), CLJA_WATERMODELING_01-09_0000033272

[83] ATSDR (2009), CLJA_WATERMODELING_01-09_0000033272

[84] NRC (2009); The 2009 NRC Report described the NRC as "organized by the National Academy of Sciences in 1916 to associate the broad community of science and technology with the Academy's purposes of furthering knowledge and advising the federal government. Functioning in accordance with general policies determined by the Academy, the Council has become the principal operating agency of both the National Academy of Sciences and the National Academy of Engineering in providing services to the government, the public, and the scientific and engineering communities." It described the National Academy of Sciences as "a private, nonprofit, self-perpetuating society of distinguished scholars engaged in scientific and engineering research, dedicated to the furtherance of science and technology and to their use for the general welfare. Upon the authority of the charter granted to it by the Congress in 1863, the Academy has a mandate that requires it to advise the federal government on scientific and technical matters."

by the NRC to address the evidence on whether adverse health outcomes are associated with past contamination of the water supply at Camp Lejeune."[85]

- "Sophisticated computer modeling techniques were used by ATSDR to make predictions about the monthly concentrations of PCE to which residents of Tarawa Terrace were exposed. To provide perspective on its estimates, ATSDR compared its monthly estimates with the U.S. Environmental Protection Agency (EPA) maximum contaminant level (MCL) for PCE in drinking water of 5 ug/L, which was established in 1985. The model estimated that starting in November 1957, the concentration of PCE delivered to residents exceeded that MCL and remained well above it until the wells were closed in 1985."[86]
- "Some of the modeling approaches used by ATSDR were 'cutting-edge,' meaning that they used computer codes and modeling techniques that are still in the research stage and have yet to be validated. Furthermore, the absence of measurement data for the first 30 years of the contamination period means the predictions, even if based on validated codes and models, cannot be evaluated for accuracy. The actual concentrations may have been higher or lower than the predictions, but that cannot be assessed. Other uncertainties were introduced into the models because assumptions had to be made about how the water system was operating. For example, little information was available on which wells were supplying water at specific time periods, so assumptions had to be made about when the contaminated wells were operating. Another uncertainty is that the models did not take into account the DNAPL form of pollutants. Given the multiple uncertainties and likely variation in contaminant concentrations, the committee concluded that the Tarawa Terrace modeling predictions should only be used to provide a general estimate of the timeframe and magnitude of exposure."[87]

The NRC recommended:

- "Because any groundwater modeling of the Hadnot Point system will be fraught with considerable difficulties and uncertainties, simpler modeling approaches should be used to assess exposures from the Hadnot Point water system. Simpler modeling will not reduce the uncertainty associated with the estimates, but they have the advantage of providing a broad picture of the timeframe and magnitude of exposure encountered by people who used water from that system more quickly and with less resources than complex modeling exercises."[88]

- **September 2011:** Dr. T. Prabhakar Clement, an NRC Committee Member and Auburn University Professor, published an article in *Groundwater* entitled "Complexities in Hindcasting Models–When Should We Say Enough is Enough?"[89]

  Dr. Clement's article echoed the NRC's concerns about the uncertainty in ATSDR's water model related to Tarawa Terrace and recommended a simpler approach for the water model related to Hadnot Point and Holcomb Boulevard to meet policy-oriented goals.

[85] NRC (2009), p. 1
[86] NRC (2009), p. 4
[87] NRC (2009), p. 4
[88] NRC (2009), p. 5
[89] Clement (2011)

- "[E]ven if one had a 'perfect' groundwater model, the final outcomes of the study would have considerable uncertainties due to lack of knowledge about actual exposures… ."[90]
- "For the CLJ project, the judgment call was made by the NRC panel, which consisted of a diverse group of 14 experts who volunteered their time to study various aspects of the problem for 2 years and prepared a report, which was reviewed by 10 external reviewers… As voluntary expert committees, such as the NRC panel, do not have any direct self-interest, their collective wisdom is likely to recommend a reasonable practical solution, although by no means it would be the perfect solution." [91]
- "The overall response to the NRC Study was mixed. Various groups of health scientists, environmental activists, one of the modeling teams, and the former CLJ residents were disappointed and severely criticized the study's conclusion that additional scientific studies cannot provide more definitive answers. In 2009, two senators from North Carolina introduced a bill to furnish hospital care, medical services, and nursing home care to veterans who were stationed at the base while the water was contaminated. In February 2010, a North Carolina congressmen introduced the *The Janey Ensminger Act* in the House of Representatives to require the Department of Veterans Affairs to provide healthcare benefits. These new policy developments directly address the healthcare needs of the community."[92]

- **January/February 2012:** ATSDR's water modeling team, led by Morris Maslia, published comments in response to Dr. Prabhakar Clement's article.

  While they disagreed with Dr. Clement's uncertainty concerns over the use of complex hindcasting models or reconstruction, ATSDR's water modeling team acknowledged the purpose of the Camp Lejeune water modeling relating to policy-oriented goals.

  - "The calibration of a model must either stand or fall on its own merits, without the benefit of future data collection that may be accomplished later in time or the lost opportunity for data collection previously foregone. At the time of calibration, when model results are provided to policy makers, a "hindcasting" model is *not* uniquely disadvantaged compared with a forecasting model just because model predictions are historical rather than latter in time. Few, if any, policy makers or the public would accept the premise that policy decisions must be delayed for several years or several decades to further validate an existing model when a decision must be forthcoming."[93]

- **August 6, 2012:** The President signs the Honoring America's Veterans and Caring for Camp Lejeune Families Act of 2012 (the Janey Ensminger Act), "*to furnish hospital care and medical services to veterans who were stationed at Camp Lejeune, North Carolina, while the water was contaminated at Camp Lejeune, to improve the provision of housing assistance to veterans and their families, and for other purposes*."[94]

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[90] Clement (2011), p. 4

[91] Clement (2011), p. 7

[92] Clement (2011), p. 8

[93] Maslia et al. (2012), p. 11

[94] 38 U.S.C. § 301 (note).

- **October 2010–March 2013:** ATSDR completed and published water modeling reports related to Hadnot Point and Holcomb Boulevard Report entitled "*Analyses and Historical Reconstruction of Groundwater Flow, Contaminant Fate and Transport, and Distribution of Drinking Water Within the Service Areas of the Hadnot Point and Holcomb Boulevard Water Treatment Plants and Vicinities, U.S. Marine Corps Base Camp Lejeune, North Carolina*," Chapters A-D.

  ATSDR's reports on Hadnot Point and Holcomb Boulevard indicated that the water modeling was intended to support an epidemiological study and not for the purpose of making exposure assessments in individuals.

  - "*The Agency for Toxic Substances and Disease Registry (ATSDR), an agency of the U.S. Department of Health and Human Services, is conducting epidemiological studies to evaluate the potential for health effects from exposures to finished water contaminated with volatile organic compounds (VOCs) at U.S. Marine Corps Base Camp Lejeune (USMCB Camp Lejeune), North Carolina*."[95]
  - "*The epidemiological studies require estimates or direct knowledge of contaminant concentrations in finished water at monthly intervals. When direct, past knowledge of contaminant concentrations in finished water is limited or unavailable, historical reconstruction is used to provide estimates of contaminant concentrations. At USMCB Camp Lejeune, historical reconstruction methods include linking materials mass balance (mixing) and water- distribution system models to groundwater-flow and contaminant fate and transport models (Maslia et al. 2007, 2009a). Results obtained from the historical reconstruction process, along with household information regarding water use and consumption, can be used in the epidemiological studies to estimate the level and duration of contaminant exposures*."[96]
  - "*ATSDR's exposure estimates cannot be used alone to determine whether you, or your family, suffered any health effects as a result of past exposure to TCE-contaminated drinking water at USMCB Camp Lejeune*."[97]

- **January 16, 2013:** ATSDR Director, Christopher Portier, sends letter to the Department of Veterans Affairs (VA) Under Secretary for Benefits, General Allison Hickey.
  - "*The Agency for Toxic Substances and Disease Registry (ATSDR) has conducted a series of environmental and epidemiologic assessments of contaminated drinking water at USMC Base Camp Lejeune. The foundation of our effort is based on modeling of the contamination of the drinking water supply before 1987. The modeling was necessary because there were relatively few drinking water samples tested for VOCs during the period of contamination; none prior to 1982, when VOC contamination was first detected*."[98]

[95] ATSDR-HP, Chapter A, p. A2

[96] ATSDR-HP, Chapter A, p. A2

[97] ATSDR-HP, Chapter A, p. A182

[98] ATSDR (2013), CLJA_WATERMODELING_01-0000076158–76159

- "*I hope this information is useful as the Department of Veterans Affairs evaluates claims from veterans who served at USMC Camp Lejeune prior to the release of our full water modeling report in the spring. ATSDR is also on schedule to release its mortality study and birth defects and childhood cancers study in spring 2013. When we finalize our water modeling and these epidemiologic studies, I will make certain that we brief the Department of Veterans Affairs staff on our findings. I would also like to recognize the efforts of your Department in supporting ATSDR's work and serving Camp Lejeune veterans and their families who were exposed to contaminated drinking water*."[99]

- **December 4, 2013:** ATSDR study entitled "Evaluation of exposure to contaminated drinking water and specific birth defects and childhood cancers at Marine Corps Base Camp Lejeune, North Carolina: a case–control study" is published in *Environmental Health*.
  - "Limited historical, contaminant-specific data were available, therefore ATSDR conducted a historical reconstruction of contaminant levels in the drinking water using groundwater fate and transport and water-distribution system models. Modeling provided monthly average estimates of the concentrations of the contaminants in drinking water delivered to residences." [100]

- **2017**: The VA "amends its adjudication regulations regarding presumptive service connection, adding certain diseases associated with contaminants present in the base water supply at U.S. Marine Corps Base Camp Lejeune (Camp Lejeune), North Carolina, from August 1, 1953, to December 31, 1987."[101]

- **2014-2024:** ATSDR conducted other epidemiological studies related to Camp Lejeune relying on the Camp Lejeune water modeling to conduct relative exposure assessments.

- **October 24, 2024:** ATSDR published a study entitled "Cancer Incidence among Marines and Navy Personnel and Civilian Workers Exposed to Industrial Solvents in Drinking Water at US Marine Corps Base Camp Lejeune: A Cohort Study" in *Environmental Health Perspectives*. Unlike past ATSDR epidemiolocal studies, this study did not rely on an exposure assessment based on ATSDR's water models related to Camp Lejeune.[102]

- **October 26, 2024:** Mr. Maslia and Dr. Aral are identified as experts for Plaintiffs in the *Camp Lejeune Justice Act* litigation.

[99] ATSDR (2013), CLJA_WATERMODELING_01-0000076158–76159

[100] ATSDR Env Health (2013), CLJA_HEAL THEFFECTS-0000165861–165879

[101] 82 FR 4173

[102] Bove (2024). Deposition, October 18, p. 20:3–11.

Section 4
Bases for Opinions

ATSDR conducted water modeling for Tarawa Terrace and subsequent water modeling for Hadnot Point and Holcomb Boulevard. For Tarawa Terrace, ATSDR stated "*ATSDR is using water-modeling techniques to provide [an] epidemiological study with quantitative estimates of monthly contaminant concentrations in finished drinking water because contaminant concentration data and exposure information are limited*."[103] For Hadnot Point and Holcomb Boulevard, ATSDR stated "*[t]he epidemiological studies require estimates or direct knowledge of contaminant concentrations in finished water at monthly intervals. When direct, past knowledge of contaminant concentrations in finished water is limited or unavailable, historical reconstruction is used to provide estimates of contaminant concentrations*."[104] Regarding uncertainty, ATSDR noted that "*[h]istorical data on the levels of contaminants in the drinking water is very limited*"[105] and "*[t]hat is why there is uncertainty and variability*."[106] ATSDR further acknowledged this uncertainty by stating that "*ATSDR's exposure assessment cannot be used to determine whether you, or your family suffered any health effects as a result of past exposure to [] contaminated drinking water at Camp Lejeune*."[107]

Mr. Maslia's report offers several opinions on the acceptability of "historical reconstruction" methods, as well as examples of past projects that utilized historical reconstruction for similar purposes to ATSDR's work at Camp Lejeune. Mr. Maslia admitted that many of the modeling projects he has worked on through his career did not use historical reconstruction. He did, however, emphasize two historical reconstruction projects in his report: Dover Township, New Jersey; and Woburn, Massachusetts.[108] However, these projects were very different in both scope and purpose from ATSDR's Camp Lejeune projects.

- Mr. Maslia testified that he was asked to apply the techniques used in the Dover Township study to Camp Lejeune.[109] In the Historical Reconstruction of the Water-Distribution System Serving the Dover Township Area, New Jersey, ATSDR's Exposure Dose Reconstruction Program modeled "*the percentage of water derived from different sources that historically supplied the water-distribution system*."[110] In other words, ATSDR estimated "*the percentage of water that a study subject might have received from each well and well field that supplied the water-distribution system*."[111] Unlike ATSDR's work at Camp Lejeune, the Dover Township study did not involve groundwater models, but rather focused on contaminant transport within the water distribution systems. Moreover, the study did not estimate

[103] ATSDR-TT, Chapter A, p. A2

[104] ATSDR-HP, Chapter A, p. A2

[105] ATSDR-TT, Chapter A, p. A99

[106] ATSDR-HP, Chapter A, p. A182

[107] ATSDR-TT, Chapter A, p. A98; ATSDR-HP, Chapter A, p. A182

[108] Maslia expert report (2024), p. 97

[109] Maslia (2024), Deposition, September 26, 91:2–16

[110] Maslia et al. (2001), p. 1

[111] Grayman et al. (2004)

concentrations of contaminants in water, but rather the "proportionate contribution" of a given well to an individual's drinking water.[112]

- In the Woburn, Massachusetts exposure study, a water distribution system model was used to simulate the percentage of a household's water that came from supply wells G and H.[113] A groundwater model was used to "*demonstrate the plausibility that contaminated water reached*" those wells, then a water distribution system model was used to assess "*the potential for a residence to receive water from wells G and H and not on actual contaminant concentration in the wells*."[114]

ATSDR's work at Camp Lejeune, therefore, is both significantly more complex and uncertain than the projects Mr. Maslia cited. Moreover, these studies were being used for a less precise purpose; here, Mr. Maslia states that ATSDR's Camp Lejeune studies may be used to show, definitively, the concentration of contaminants to which a given individual was exposed.[115] This is not what the ATSDR reports themselves say the purpose of the study was, and it is not what was done at Dover Township or Woburn.

For example, to calculate the estimated monthly contaminant concentrations, ATSDR implemented complex calculation methodologies. As Mr. Maslia indicated in his 2010 deposition, "*we used, I believe, more sophisticated methods*."[116] When asked whether these methods were so sophisticated as to be novel and unreliable, Mr. Maslia indicated that these methods were "*[n]ot unreliable. Novel application, yes*."[117] Mr. Maslia provided a more detailed description of the application of these methods: "*[w]e were predicting -- or reconstructing backwards in time for 30, 35 years at a monthly interval, which is a -- from a groundwater modeling standpoint, a fairly fine timeline, typically. And in terms of, say, remediation practices where they use these similar models, you may look at years – or five – of years trying to clean up.*"[118] He also highlighted that "*you do not necessarily see published results in terms of monthly values. So that was a very refined time step in terms of a groundwork model.*"[119] Mr. Maslia characterized this by saying "*[s]o from that standpoint, that's probably, you know, edge of the envelope of what's been done*."[120]

Mr. Maslia was deposed again in 2024 and commented on his 2010 statements, saying that these methods allowed ATSDR "*to go backwards in time, reconstruct based on either available data in the 1980s or current day information*."[121] Mr. Maslia continued his description of the calculation approach by indicating that *"[m]any modeling remediation-type studies collect field data present day and then, of course, project forward in time, but this was a unique application of -- of going backwards in time.*"[122]

In his expert report, Mr. Maslia provided examples of studies that ATSDR conducted as part of its Exposure-Dose Reconstruction Program.[123] Mr. Maslia indicated that *"[t]he overall goal of the Exposure-Dose Reconstruction Program (EDRP) was to enhance the agency's capacity to characterize exposure and*

[112] Maslia et al., 2001, p. 2—4.

[113] Lagakos et al. (1986), p. 585.

[114] Costas et al. (2002), p. 2–4.

[115] Maslia (2024), Expert Report, p. 97

[116] Maslia (2010), Deposition, June 30, p. 45

[117] Maslia (2010), Deposition, June 30, p. 45

[118] Maslia (2010), Deposition, June 30, p. 45

[119] Maslia (2010), Deposition, June 30, p. 45-46

[120] Maslia (2010), Deposition, June 30, p. 46

[121] Maslia (2024), Deposition, September 26, p. 216

[122] Maslia (2024), Deposition, September 26, p. 216

[123] Maslia (2024), Expert Report, p. 12

dose to better support health assessments and consultations, health studies, and exposure registries."[124] It is important to view such studies from the standpoint of (a) whether calculations are performed for hindcasting past conditions or predicting future conditions; and (b) how the results of these studies are used, e.g. remedial design versus, for example, health effects. The uncertainty of the model results can be significant and its impact on decision making can be substantial.

When predictive calculations are performed, the associated uncertainty of these calculations may be high or very high, depending on assumptions regarding the parameters used in the corresponding models. These uncertainties can be reduced as more data become available and the model can be further constrained. However, this is not possible when calculations are used for hindcasting. In that instance, the historical data available cannot be further augmented. Things become more complicated when available data are limited or non-existent. Under these circumstances, developing calculation tools requires relying heavily on professional judgment. As will be shown below, professional judgment and expert knowledge cannot replace site-specific data that should inform parameter values in the models, or lack of observed data that should constrain the model calibration. Without these two pieces of information, "novel" or "edge of envelope" approaches can be very complex, incorporating professional judgment for interpreting aquifer and chemical characteristics, but they cannot overcome the inherent limitations associated with the lack of data. In that sense, they can even be considered speculative and unfounded.

Parameter estimation methods and programs have been developed over the last three decades to assist the complex process of calibrating models to observed data.[125] Numerous studies have been published and conferences held on this subject.[126] However, over the last two decades, the focus of scientific research has shifted from calibration to uncertainty analyses, recognizing the fact that "*environmental models are built to make predictions that support the making of important management decisions. These predictions are often accompanied by a large amount of uncertainty – uncertainty that must be accommodated in any sensible decision-making strategy. Quantification of uncertainty allows evaluation of the risks associated with different management strategies*."[127] In this context, "predictions" refer to the model output, regardless of whether its results are used for hindcasting or forecasting, as the uncertainty analysis investigates the non-uniqueness of the model solution.[128]

In recent years, advanced uncertainty analysis methods have been developed to take advantage of computational capabilities provided by model computers.[129] Advanced uncertainty methods recognize the need for "*the development of means to explore calibration-constrained predictive uncertainty*."[130] For the uncertainty of a model prediction, critical to the variability of model parameters is "*the extent to which this variability is constrained by the necessity for model outputs to respect historical measurements of system state.*"[131] In other words, the entire discussion about model calibration and uncertainty is founded on the concept of *respecting historical measurements*. When historical measurements are not available, uncertainty bounds rely solely on professional judgment. As will be shown in the discussion below, historical and site-specific data can disprove assumptions based on professional judgment and expert knowledge. As a result, actual conditions can vary significantly from conditions assumed based on

[124] Maslia (2024), Expert Report, p. 12

[125] Doherty (2015), p.1

[126] 2023 PEST Conference – The Path from Data to Decisions, March 5-10, 2023, La Jolla, CA

[127] Doherty (2015), p.1

[128] Anderson et al. (2015), p. 378

[129] Anderson et al. (2015), p. 469

[130] Doherty (2015), p.2

[131] Doherty (2015), p.2

professional judgment. Advanced uncertainty methods that specifically consider calibration datasets have become the prevailing standard in recent years due to their holistic approach to uncertainty.[132,133]

The Pacific Northwest National Laboratory (PNNL) conducted an analysis to address issues related to hydrologic uncertainty and its impacts on dose from contaminated sites and waste disposal facilities. The impact of ignoring site-specific data was highlighted: *"[t]he complex model assumed the availability of an extensive dataset on which to base the random field characterization of the subsurface. Uncertainty in predicted dose was correspondingly small, with the peak dose coefficient of variation being 30%. When the variances of parameters in the simplified model were based on a generic dataset, the uncertainty in predicted peak dose was much larger; the coefficient of variation was 52% in this case. When the variances of parameters in the simplified model were based on the available site-specific data, the coefficient of variation for the peak dose was reduced to 22%*."[134] In other words, generic datasets for model parameters, even when selected based on professional judgment and expert knowledge, can fail to properly quantify uncertainty, compared to considering site-specific data. The resulting uncertainty can be much higher when generic data are used for constructing the model.

Examples of use of models include the evaluation of the nature and extent of contamination and/or design of a system for containing a contaminant plume, aquifer restoration to certain cleanup standards, or evaluation of ultimate fate and transport of a contaminant plume.[135] Model uncertainty may have significant impacts on the design of plume containment or aquifer restoration systems, as simulation results may substantially under- or over-estimate plume migration patterns and aquifer response to pumping. This type of model failure can be mitigated by updating model calibration via collection of more data during the remediation phase and revising model predictions. It can also be mitigated by expanding the remedial design.

However, when models are used for hindcasting or forecasting conditions that are directly translated to substantially more important decisions, such as health impacts, the implications of model uncertainty have to be viewed more critically. Camp Lejeune is a suitable case in point. ATSDR reconstructed historical conditions at Camp Lejeune to calculate how much contamination (i.e., dose)[136] people at Camp Lejeune were exposed to, by implementing "*a unique application of -- of going backwards in time,"*[137] and "*reconstructing backwards in time for 30, 35 years at a monthly interval*,"[138] using "*[n]ovel application*"[139] of significant complexity.

In his expert report, Dr. Aral concurred with the following statement by Dr. Robert Clark, Chair of the Expert Review Panel for the ATSDR Camp Lejeune studies: "*[f]rom a scientific viewpoint it would be ideal to have independent datasets. One set could be used to calibrate the models, and the second data set used for validation. If one is developing a model based on experimental data this approach can be built into the combined experimental and modeling effort. However, it has been my experience that such an ideal situation rarely exists in "real world" situations. Therefore, in my opinion, the best approach is to use the available datasets in conjunction with sound engineering principles and the investigator's best judgment to establish the validity of the exposure models.*"[140] Dr. Aral then stated that "*I concur with Dr. Clark's*

[132] Tonkin and Doherty (2009)

[133] Fienen et al. (2006)

[134] Meyer and Orr (2002), p. 43

[135] Ahlfeld et al. (1988); Wagner (1995)

[136] ATSDR-TT, Chapter A, p. A98

[137] Maslia (2024), Deposition, September 26, p. 216

[138] Maslia (2010), Deposition, June 30, p. 46

[139] Maslia (2010), Deposition, June 30, p. 45

[140] Aral (2024), Expert Report, p. 48

assessment given above. It is my opinion that ATSDR used the best available datasets, sound science and engineering principles, and professional judgment to establish the best possible reconstructed values of historical contaminant concentrations, and that, within a reasonable degree of scientific and engineering certainty, these were the contaminant levels delivered to Tarawa Terrace, Hadnot Point, and Holcomb Boulevard."[141] Dr. Clark indicated that ideal conditions rarely exist in the "real world" and that "available datasets" should be used. In Camp Lejeune, "available datasets" are practically non-existent. In fact, there are little to no site-specific data for key modeling parameters, historical operational data for water-supply wells, or, more importantly, observed data to use for constructing and calibrating the model. It is not obvious what Dr. Aral referred to when he stated that "*ATSDR used the best available datasets."* The reality is that ATSDR constructed a model based almost exclusively on professional judgment and assumptions that cannot be tested.

Dr. Clement, a professor in the Department of Civil Engineering at Auburn University, published an issue paper in the journal *Groundwater* in 2011, expressing his opinions regarding ATSDR's modeling approaches and methods in Tarawa Terrace.[142] He commented on ATSDR's uncertainty analysis, indicating that "*the results appear to be reasonable because the Monte Carlo simulations indicated a narrow band within which 95% of the model-simulation results resided. The figure shows that 95% confidence band becomes narrower as we move back from the 1980's (where there is no data); this implies that the groundwater model was able to make confident hindcasts for the 1950s and 1960s even if there are no past data to calibrate the model. The figure also shows that closer to the initial starting point the confidence band is almost 100%, implying that our knowledge of initial conditions, initial source loadings, and initial stresses is almost exact."* In his comments, Dr. Clement highlighted the shortcomings of the uncertainty analysis conducted by ATSDR. He indicated that ATSDR's analysis implied almost exact knowledge of past conditions. But that would be impossible, given that there were no past data to calibrate the model, and there was no confidence in the assumptions about the history (i.e., variability in timing and magnitude) of the contaminant contributions from the identified source, or generally in the aquifer conditions.

The NRC, in its review of the scientific evidence on water contamination at Camp Lejeune,noted that "*the absence of measurement data for the first 30 years of the contamination period means the predictions, even if based on validated codes and models, cannot be evaluated for accuracy*."[143] The NRC further stated, regarding the model challenges and limitations for Tarawa Terrace, that "*[a]lthough ATSDR recognized and tried to account for the limitations and uncertainties associated with its models, the committee judges that—because of the sparse set of water-quality measurements, the need to make unverifiable assumptions, and the complex nature of the PCE source—it is virtually impossible to estimate exposure to historical levels of PCE and its degradation products accurately. Reporting precise values based on model predictions gives the misleading impression that the exposure of the former residents and workers at Tarawa Terrace during specific periods can be accurately defined.*"[144] For Hadnot Point, the NRC indicated that "*any groundwater modeling of the Hadnot Point system will be fraught with considerable difficulties and uncertainties.*"[145]

The NRC also opined on ATSDR's use of complex calculation approaches in the absence of historical observed data: "*Even with the use of reasonable and, in some cases, advanced approaches, limitations in data availability and quality cannot be overcome*."[146]

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[141] Aral (2024), Expert Report, p. 48

[142] Clement (2011)

[143] NRC (2009), p. 4

[144] NRC (2009), p. 16

[145] NRC (2009), p. 5

[146] NRC (2009), p. 22

Dr. Aral also states in his expert report: "*[t]he NRC committee should accept the fact that answers to uncertainty questions cannot be answered "accurately.*"[147] Dr. Aral continued, saying that: "*Our uncertainty analyses are not provided to give "accurate" answers to the problem studied. Instead, our uncertainty analyses are used as estimates that would indicate the variability range of deterministic results provided earlier."*[148] In response to the NRC's comment about "*some important limitations in ATSDR's modeling efforts because of the sparse set of water quality measurements, the need to make unverifiable assumptions, and the complex nature of the PCE source contamination,*"[149] Dr. Aral admitted that there are limitations to the modeling analyses conducted by ATSDR.[150] He also suggested that the level of detail of the uncertainty analyses conducted by ATSDR enveloped the effect of those uncertainties.[151] And he concluded by saying that "*an uncertainty that can be verified would no longer be uncertain*."[152]

However, as discussed above, model accuracy is key to a model that is used as an important decision-making tool. When no site-specific data or historical observed data are available, the model cannot be assessed for its accuracy, and the same is true about its uncertainty. As will be shown below, the ATSDR model outputs can vary significantly when simple corrections to key model parameters are made, or the uncertainty of important factors (such as the source release history) are considered. However, these outputs are outside the realm of uncertainty determined by ATSDR in their analyses. This is evidence of the fact that a process can be complex, and expert knowledge and professional judgment can be imparted in the analysis, but neither can substitute for observed data.

In Summary (Opinion 1): ATSDR constructed models for historical reconstruction. To construct these models, they combined complex processes and methods. However, these models were largely not constructed using site-specific data or calibrated to observed data for the first 30 years of simulation. ATSDR relied extensively on professional judgment for constructing these models. Despite the extensive assumptions and substantial uncertainties underpinning these models, ATSDR used the models to calculate monthly estimates of concentrations in the water-supply wells and the water treatment plant. ATSDR's uncertainty analyses did not assess model accuracy, as there were no observed data to support such assessment. Thus, model accuracy was replaced by process complexity and professional judgment.

Below are detailed discussions of the bases for my remaining opinions, provided separately for the Tarawa Terrace and Hadnot Point-Holcomb Boulevard areas.

In what follows, an important clarification is necessary regarding the historical reconstruction of contaminant concentrations at the WTPs. ATSDR stated that "*for this study, finished water is defined as groundwater that has undergone treatment at a water treatment plant and was subsequently delivered to a family housing unit or other facility. Throughout this report and the Hadnot Point–Holcomb Boulevard report series, the term finished water is used in place of terms such as finished drinking water, drinking water, treated water, or tap water*."[153] However, ATSDR used simulated contaminant concentrations in the influent to the WTP to estimate concentrations in the water delivered to a family housing or other facility. In this process, ATSDR ignored any contaminant losses that would occur during treatment. This was an important assumption of significant impacts, as discussed in the expert report of Dr. Hennet (2024). Therefore, references to historical reconstruction of VOC concentrations hereafter are associated with

[147] Aral (2024), Expert Report, p. 56

[148] Aral (2024), Expert Report, p. 56

[149] Aral (2024), Expert Report, p. 56

[150] Aral (2024), Expert Report, p. 56

[151] Aral (2024), Expert Report, p. 56

[152] Aral (2024), Expert Report, p. 56

[153] ATSDR, Chapter A, Supplement 6, p. S6.21

concentrations in the influent to the treatment plant, and not after-treatment "finished" water that entered the water distribution network.

4.1 Tarawa Terrace

Water quality samples taken at Camp Lejeune in the 1980s revealed contamination of VOCs. Because there were no water quality samples analyzed for the VOCs addressed in Mr. Maslia's and Dr. Aral's reports prior to the 1980s, ATSDR attempted to use mathematical modeling to reconstruct historical concentrations of these contaminants in water supply wells and at the WTPs in the absence of measured data.

ATSDR used a model to reconstruct groundwater flow and contaminant transport at Tarawa Terrace family housing. In creating their conceptual model, ATSDR relied on past investigations of hydrogeologic conditions in the aquifer below Camp Lejeune. Using the conceptual model, ATSDR created a groundwater flow model, using MODFLOW. The groundwater flow model was created with limited available data. The groundwater flow model was then used to create a contaminant fate and transport model, using MT3DMS, which also relied on limited data. Next, ATSDR performed model calibration, sensitivity analysis, and uncertainty analysis.

Based on my professional judgment, there were insufficient data to conduct reliable model calibration and uncertainty analysis. Given the fact that prior to 1982, no water quality data were available, ATSDR's model was highly uncertain. ATSDR's uncertainty analysis evaluated a range of parameter values, some of which, when compared to site specific data, did not reflect the site conditions. An uncertainty analysis should provide a range of potential model outcomes that envelops the calibrated model. The calibrated model should generally sit in the middle of the uncertainty range. However, ATSDR's calibrated model sits at the top of the uncertainty range, especially for approximately the first ten years of the simulation timeframe. This demonstrates that the calibrated model was biased high.

Moreover, as discussed below, ATSDR improperly characterized the PCE source release date and overestimated the gradual mass loading into the aquifer from that source.

ATSDR developed a second model, TechFlowMP, to simulate (a) the presence of PCE in both gas and water, for considering volatilization of PCE in the unsaturated zone, and (b) the migration of PCE degradation by-products in groundwater. Unlike TechFlowMP, MT3DMS could only simulate PCE concentrations in groundwater. TechFlowMP calculated PCE concentrations in groundwater that were lower than those calculated using MT3DMS, because of inconsistent implementation of the contaminant source term.

The NRC highlighted the model challenges and limitations: "*Although ATSDR recognized and tried to account for the limitations and uncertainties associated with its models, the committee judges that—because of the sparse set of water-quality measurements, the need to make unverifiable assumptions, and the complex nature of the PCE source—it is virtually impossible to estimate exposure to historical levels of PCE and its degradation products accurately. Reporting precise values based on model predictions gives the misleading impression that the exposure of the former residents and workers at Tarawa Terrace during specific periods can be accurately defined.*"[154]

The NRC also opined on ATSDR's use of complex calculation approaches in the absence of historical observed data: "*Even with the use of reasonable and, in some cases, advanced approaches,*

[154] NRC (2009), p. 16

limitations in data availability and quality cannot be overcome."[155] Dr. Waddill indicated his agreement with this conclusion in his deposition.[156] I concur with this conclusion.

As part of this litigation, Dr. Jones and Mr. Davis conducted a post-audit, which considered data from the site remediation and extended the model to 2008. This extended model consistently overestimated concentrations of contaminants in groundwater compared to observed values. This demonstrates that ATSDR's Tarawa Terrace's calibrated model resulted in concentrations higher than those observed in the aquifer.

4.1.1 Available Data are Limited or Non-Existent

To create the groundwater flow model, ATSDR used available data, which included:

- Horizontal hydraulic conductivities from 36 aquifer test analyses at Tarawa Terrace and adjacent areas;[157]
- Aquifer-specific yield and storativity values, computed from four aquifer tests in the vicinity of Tarawa Terrace;[158] and
- Precipitation data from the Maysville-Hoffman Forest station with records from 1951-1994.[159]

Pumpage information at individual supply wells were not available for the study period. ATSDR developed assumed well pumping schedules and flow rates through a complex process. To do that, they relied on ancillary data, including well-capacity data and average water supply demand for the TT WTP for different periods (i.e., sparse data).[160]

All other model parameters were based on a literature review and the professional judgment of the modelers.

Limited data were available on the actual operation of water supply wells. In the absence of documentation of historical water supply well operations, ATSDR modeled a hypothetical well pumping schedule, which was used in the groundwater flow model.

To construct the contaminant transport model, ATSDR used model parameters that were based on a literature review and the professional judgment of the modelers.

Model calibration was based on:

- Contaminant concentration data at water supply wells from 1984-1985 and 1991;[161] and
- Contaminant concentration data at monitoring wells from previous remediation investigations.[162]

Although operations at ABC One-Hour Cleaners started sometime in 1954,[163] sampling data from water supply wells were not available before January 1985. This means there was a thirty-year period for which there is no historical water quality data that could be used to inform the model calibration. Appendix D lists all the sampling data from the water supply wells and WTP available to ATSDR.

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[155] NRC (2009), p. 22

[156] Waddill (2024), Deposition, August 6, p. 124:5

[157] ATSDR-TT, Chapter C, p. C14

[158] ATSDR-TT, Chapter C, p. C40

[159] ATSDR-TT, Chapter C, p. C21

[160] ATSDR-TT, Chapter C, p. C22

[161] ATSDR-TT, Chapter E, p. E4

[162] ATSDR-TT, Chapter E, pp. E5-E23

[163] Brigham (2024), Expert Report

No measured data or information was available on the mass loading from the contaminant source. ATSDR relied on information on solvents used by ABC One-Hour Cleaners and site investigations of the Superfund site in the 1990s. ATSDR relied on this information to estimate contaminant mass loading into the aquifer, which was assumed to be constant from 1953 to 1984. ATSDR increased its initial estimate of mass loading to a much higher value during model calibration.

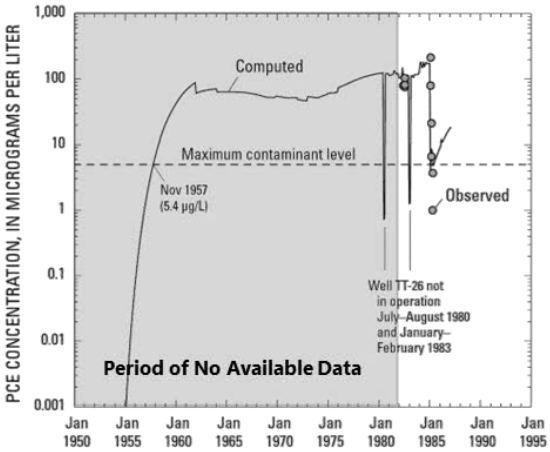

Figure F27. Computed and observed concentrations of tetrachloroethylene (PCE) in finished water at the water treatment plant, Tarawa Terrace, U.S. Marine Corps Base Camp Lejeune, North Carolina, January 1951–December 1984. [µg/L, microgram per liter]

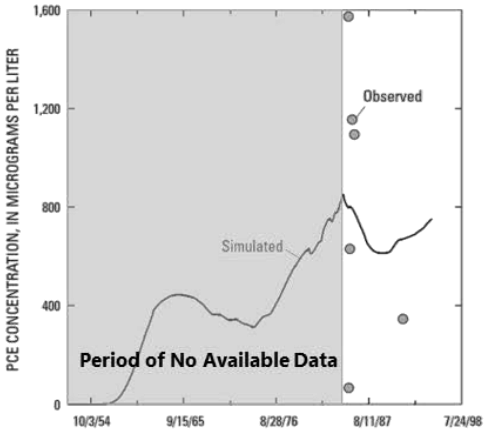

Figure F16. Simulated and observed tetrachloroethylene (PCE) concentrations at water-supply well TT-26, Tarawa Terrace, U.S. Marine Corps Base Camp Lejeune, North Carolina, January 1952–December 1994 (see Figure F6 for location).

Figure 5: Period of No Available Data for ATSDR's Model Reconstruction in Tarawa Terrace

In Summary (Opinion 2): All of this is to say that the data ATSDR had to inform its modeling for reconstructing thirty years of historical aquifer conditions were extremely limited. Data that were available only existed for short time periods mainly between 1982 and 1985. As illustrated by the yellow highlighted area in Figure 5,[164] ATSDR calibrated a model with limited to no data to constrain that calibration. In some sense, this is like fitting a line to a point. Even after expert knowledge is applied, there are multiple configurations of model outputs that could fit the observed data, just as there are multiple ways to fit a line to a point. Although ATSDR attempted to quantify that uncertainty, certain parameter ranges they used in their uncertainty analysis did not encompass the full range of values known from site specific data, resulting in biased high estimates.

4.1.2 ATSDR's Model Calibration was Based on Limited Data and was Biased High

ATSDR conducted its calibration of the TT groundwater flow model relying on a limited dataset. According to ATSDR, *"[h]istorical water level data were mainly unavailable prior to 1978, with the exception of one or two measurements at the time of construction of several wells*."[165] Furthermore, "*[w]ell construction data also were somewhat limited and possibly affected the assignment of pumpage to model layers*."[166] Hence, monthly pumping rates of all wells in the model were developed using a complex reconstruction process based on assumptions. These rates were fixed but uncertain, and underpinned model performance.

[164] ATSDR-TT, Chapter F, pp. F34 and F43

[165] ATSDR-TT, Chapter C, p. C38

[166] ATSDR-TT, Chapter C, p. C39

As ATSDR admitted, "*for the most part, simulation results are unqualified for the years 1951-1977, based on comparisons of observed and simulated water levels.*"[167]

The contaminant transport model was constructed based on numerous assumptions on parameter values as site specific data were limited or nonexistent. Also, the timing of releases from the source at ABC One-Hour Cleaners and the magnitude of its contributions to contamination were uncertain. This means that there are legitimate questions about whether ATSDR's model reconstruction of historical concentrations is accurate. This is especially true considering that ATSDR admitted that "*simulated PCE concentrations moderately to substantially overpredicted observed concentrations at water-supply wells*."[168] Mr. Maslia acknowledged "it overpredicts" in his 2024 deposition.[169]

For its model calibration, ATSDR considered 36 observed PCE concentrations at water-supply wells from samples collected at different times in 1985 (29 samples) and July 1991 (7 samples). ATSDR considered a calibration target range of "+/- one-half order of magnitude range" of the observed concentration.[170] Figure 6 shows ATSDR's Table F13, with its model calibration results, including simulated and observed PCE concentrations at the water supply wells, and the corresponding calibrated target range for each well.

[167] ATSDR-TT, Chapter C, p. C38

[168] ATSDR-TT, Chapter F, p. F33

[169] Maslia (2024), Deposition, September 26, p. 228.

[170] ATSDR-TT, Chapter F, p. F32

Table F13. Simulated and observed tetrachloroethylene (PCE) concentrations at water-supply wells and calibration target range, Tarawa Terrace and vicinity, U.S. Marine Corps Base Camp Lejeune, North Carolina.

[µg/L, microgram per liter; ND, not detected; J, estimated]

Site name	Date	PCE concentration, in µg/L		Calibrated target range, in µg/L
		Observed	Simulated	
[1]RW1	7/12/1991	ND	0.0	0.0–2.0
[1]RW2	7/12/1991	760	1,804	240–2,403
[1]RW3	7/12/1991	ND	0.0	0.0–2.0
[2]TT-23	1/16/1985	132	254	41.7–417
	2/12/1985	37.0	254	11.7–117
	2/19/1985	26.2	253	8.3–82.8
	2/19/1985	ND	253	0.0–10.0
	3/11/1985	14.9	253	4.7–47.1
	3/11/1985	16.0	253	5.2–52.5
	3/12/1985	40.6	253	12.8–128
	3/12/1985	48.0	253	15.4–154
	4/9/1985	ND	265	0.0–2.0
	9/25/1985	4.0	279	0.3–12.6
	7/11/1991	ND	193	0.0–5.0
[2]TT-25	2/5/1985	ND	6.2	0.0–10.0
	4/9/1985	ND	8.6	0.0–2.0
	9/25/1985	0.43J	18.1	0.14–1.4
	10/29/1985	ND	20.4	0.0–10.0
	11/4/1985	ND	20.4	0.0–10.0
	11/13/1985	ND	20.4	0.0–10.0
	12/3/1985	ND	22.8	0.0–10.0
	7/11/1991	23.0	72.6	7.3–72.7
[2]TT-26	1/16/1985	1,580	804	500–5,000
	2/12/1985	3.8	804	1.2–12
	2/19/1985	55.2	798	17.5–175
	2/19/1985	64.0	798	20.2–202
	4/9/1985	630	801	199–1,999
	6/24/1985	1,160	732	367–3,668
	9/25/1985	1,100	788	348–3,478
	7/11/1991	340	670	111–1,107
[2]TT-30	2/6/1985	ND	0.0	0.0–10.0
[2]TT-31	2/6/1985	ND	0.15	0.0–10.0
[2]TT-52	2/6/1985	ND	0.0	0.0–10.0
[2]TT-54	2/6/1985	ND	5.8	0.0–10.0
	7/11/1991	ND	30.4	0.0–5.0
[2]TT-67	2/6/1985	ND	3.9	0.0–10.0

[1]See Figure F6 for location

[2]See Figure F1 for location

Note: Calibration target ranges for analyses listed as not detected are detection limits noted in Table F2

Figure 6: Simulated and Observed PCE Concentrations at Water Supply Wells in Tarawa Terrace (ATSDR, Table F13, Chapter F)

According to ATSDR "*[o]f the total of 36 comparisons of simulated to observed PCE concentrations in all water-supply wells used to calibrate the Tarawa Terrace fate and transport model (Table F13), including 'non-detected' results, 17 comparisons or 47 percent conformed to the calibration standard, and 19 comparisons or 53 percent violated the standard*."[171] ATSDR also calculated the geometric bias of the calibrated model, which is another type of metric for comparing simulated and observed values.[172] A geometric bias of 1.0 indicates perfect agreement between the two sets of values. ATSDR calculated the geometric bias of its calibrated model for the 19 pairs of values corresponding to detection (i.e., excluding non-detections).[173] The calculated geometric bias was 5.9, much higher than 1.0, indicating a biased-high calibrated model.

A statistical evaluation of the contaminant transport model calibration for the entire historical period could not be performed as observed data were not available prior to 1985. Hence, the resulting model, which describes thirty years of aquifer conditions beginning in 1951 and attempts to reconstruct potential contamination in the aquifer during those years, was calibrated to limited water-quality data available only after 1985. The calibrated model was biased high.

In his 2011 paper in *Groundwater*, [174] Dr. Clement expressed concerns about the fact that the ATSDR "*model was calibrated to limited number of datapoints*." Maslia et al., in their editorial response to Dr. Clement's paper in the same journal in 2012,[175] indicated that ATSDR completed a four-stage calibration process. However, they never addressed the main point of Dr. Clement's opinion regarding the lack of historical water-quality data for model calibration.

In Summary (Opinion 2): ATSDR's model calibration did not rely on observed data prior to 1984. In fact, the model was calibrated to a very limited dataset even after 1984. Calibration statistics indicate that estimated monthly contaminant concentrations were conservative and biased-high.

In this section, I focus on certain assumptions and parameters due to their significant impact on the model results. It should be noted that this discussion is not intended to be inclusive of all assumptions or parameters I believe were inappropriately selected.

4.1.2.1 The PCE Source Release Start Date at ABC One-Hour Cleaners Was Incorrect.

In its contaminant transport model, ATSDR represented the PCE contamination source at Tarawa Terrace as ABC One-Hour Cleaners. ATSDR used a constant mass loading term, which means that the same amount of contaminant mass was modeled to enter the groundwater every day, for the entire period this source was in place. ATSDR assumed the source was active from January 1, 1953, to December 31, 1984. However, as discussed extensively in the expert report of Dr. Brigham, this start date is likely incorrect. According to Dr. Brigham, ABC One-Hour Cleaners opened in June 1954. Based on my review of Dr. Brigham's expert report, July 1, 1954, is a more appropriate start date for the release of PCE into the soil at ABC One-Hour Cleaners. The impact of this discrepancy in release start dates is that the PCE plume reached the water supply wells sooner in ATSDR's model.

In Summary (Opinion 2): ATSDR's model was constructed based on incorrect information regarding the start date of contaminant releases from ABC One-Hour Cleaners. This incorrect assumption resulted in estimated monthly contaminant concentrations that were conservative and biased-high in the early 1950s.

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[171] ATSDR-TT, Chapter F, p. F33

[172] Geometric bias is the mean value of the logarithmic ratios of simulated to observed values

[173] ATSDR-TT, Chapter F, p. F33

[174] Clement (2011)

[175] Maslia et al. (2012)

4.1.2.2 Selected Geochemical Parameters Were Incorrect

As discussed in the expert report of Dr. Hennet (2024), one of the most important parameters in characterizing the rate of migration of a contaminant plume is its retardation. If a contaminant plume's migration is not retarded relative to groundwater flow, the retardation factor is equal to 1, and the contaminant moves at the same velocity as the groundwater. When a contaminant's velocity is slower than groundwater , the retardation factor is greater than 1. The retardation factor is calculated using the distribution coefficient of the contaminant (K_d), the dry bulk density of the aquifer (ρ_b), and the total porosity of the aquifer (n). Wiedemeier et al. (1999) provide a more extensive discussion on the calculation of the retardation factor.[176]

ATSDR miscalculated one of the parameters in the retardation factor, the bulk density. Below is a discussion on how the components of the retardation factor were defined by ATSDR and where the error was made.

4.1.2.2.1 Bulk Density Value was Incorrect

Data for the dry bulk density of the aquifer were not available. To estimate the dry bulk density, the specific gravity (G_s) of the soil, the density of water (ρ_w), and the aquifer total porosity (n) are required. ATSDR referred to literature sources for ranges of these properties that are appropriate for soils encountered at Camp Lejeune. Morris and Johnson (1967) analyzed hundreds of sand, clay, and silt samples, and reported associated ranges of specific gravity for each group. Based on these ranges, ATSDR assigned a specific gravity of 2.7 to all sediments represented by the seven layers of the MT3DMS model. The density of water is equal to 1,000 g/cm^3. Total porosity estimated from two samples collected in the vicinity of ABC One-Hour Cleaners was 32.9% for the clayey sand, and 36.5% for the silty sand.

To calculate the bulk density, the following formula is used:[177]

$$\rho_b = \frac{Gs \times \rho_w}{1 + \frac{n}{1-n}}$$

ATSDR calculated the bulk density of the aquifer to be 170 pounds per cubic foot (lbs/ft^3) or 77,100 grams per cubic foot (g/ft^3). When this value is converted to g/cm^3, it is equal to 2.7 g/cm^3. This number is not characteristic of soils at Camp Lejeune, as ATSDR later indicated in their analysis for Hadnot Point.[178] Closer inspection of the calculated value indicates that ATSDR did not apply this formula in its entirety; the denominator in this fraction was never included in the calculation. As a result, the calculated bulk density was much higher than it should have been. This has a significant impact on the calculation of the retardation factor, resulting in faster (sooner) arrival of PCE at the water-supply wells, as will be described in Section 4.1.3.2. In the Hadnot Point model, this error was not repeated. ATSDR used a value of 1.65 g/cm^3, or 46,700 g/ft^3, as was found in the literature for soils in Camp Lejeune.[179]

4.1.2.2.2 Distribution Coefficient (K_d) Value was Out of Range and Biased Low

ATSDR indicated that no site-specific data were available for estimating the K_d. For that reason, they referred to literature sources for K_d values, for soils similar to those encountered at Camp Lejeune.[180] According to this literature source, the range of K_d values was:

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[176] Wiedemeier et al. (1999), p. 142-43

[177] Wiedemeier et al. (1999), p. 142-43

[178] ATSDR-TT, Chapter A, Supplement 6, p. S6.14

[179] ATSDR-HP, Chapter A, Supplement 6, p. S6.14

[180] ATSDR-TT, Chapter F, p. F28

- 0.25-0.76 mL/g, averaging 0.39 mL/g, for sands;
- 0.21-0.71 mL/g, averaging 0.40 mL/g, for silts; and

ATSDR used a mean value of 0.40 mL/g as the starting value in the model calibration process. However, the final K_d value, determined by ATSDR through model calibration, was 0.14 mL/g. This value is out of range for the soils in the aquifer at Camp Lejeune. In addition, this value was more than two times lower than the value used by ATSDR for PCE in the Hadnot Point model, despite ATSDR's statement that the same aquifers are encountered in both models.[181] ATSDR ultimately selected a K_d value of 0.30 mL/g for PCE, through "*refinement during the model calibration process*."[182]

The reason for this significant drop in the K_d value estimated by ATSDR during model calibration is the erroneous value for the bulk density that ATSDR calculated, as discussed above.

4.1.2.3 ATSDR's K_d Adjustments During Model Calibration were Biased Low

To compensate for the erroneously calculated bulk density, ATSDR's calibrated K_d value was below the range that ATSDR considered reasonable for the soil types at Camp Lejeune. Note that ATSDR did not make the same mistake with the bulk density in the Hadnot Point model, meaning they corrected the bulk density and K_d values later. However, they did not return to the Tarawa Terrace model to make adjustments based on this knowledge.

ATSDR started the calibration process using parameter values that were based on assumptions considering limited data and, mainly, literature sources. As mentioned above, one of the parameters that was adjusted during model calibration was the distribution coefficient (K_d). The starting value of 0.40 mL/g corresponded to a retardation factor of 6.44.[183] This starting K_d value was reduced by a factor of 3, to 0.14 mL/g, for a retardation factor of 2.93.

If ATSDR had used a retardation factor of 6.44, corresponding to the initial K_d value of 0.40 mL/g, this would have resulted in slower migration of the PCE plume that originated from ABC One-Hour Cleaners and reached well TT-26 first. It would also have resulted in lower concentrations at well TT-26 as more mass would have been sorbed by the soils. However, ATSDR selected the much lower K_d value and, therefore, retardation rate, resulting in a PCE plume that arrived sooner at well TT-26 than it would have if a higher retardation rate were selected. As a result, ATSDR estimated that there were higher PCE concentrations in the influent to the WTP at earlier times than there would have been if the higher retardation factor were selected.

In Summary (Opinions 2 and 3): ATSDR's model was constructed based on parameter values that were either incorrect (bulk density) or out of range (K_d). Parameter values were not based on site-specific data but obtained from literature sources. In addition, parameter values in the Tarawa

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[181] "Because field data describing contaminant fate and transport parameters is lacking for the HPHB study area and the TT study area is adjacent to the HPHB study area, the probability density functions described by Maslia et al. (2009) were used to generate a range of transport parameters values for the analyses reported herein." ATSDR-TT, Chapter A, Supplement 6, p. S6.34

[182] ATSDR assumed an f_{oc} value of 0.002, and a range of K_{oc} values from the literature (ATSDR-TT, Chapter A, Supplement 6, Table S6.4, p. S6.14). In that same table, ATSDR provided the corresponding calculated range of K_d values for PCE, varying between 0.03 and 21.43 mL/g (the K_d numerical value is the same when units of L/kg are used).

[183] ATSDR considered values of effective porosity from literature sources for soils similar to those in Camp Lejeune. The analysis of soil samples from the literature source cited by ATSDR provided the basis for the estimates ATSDR considered. According to this analysis, and for the fine silty and clayey sands in the aquifer system in Camp Lejeune, the effective porosity is about 20%, which is the value used by ATSDR in the model (ATSDR-TT, Chapter F, p. F28)

Terrace model were different than those used in the Hadnot Point model, even though both models simulated similar hydrogeologic conditions. These incorrect assumptions resulted in faster plume migration in the aquifer and estimated monthly contaminant concentrations that were conservative and biased-high.

4.1.2.4 Site-Specific Data for Calculating K_d Resulted in Higher K_d Value

As mentioned earlier, ATSDR considered reported values from literature sources for estimating a K_d value for PCE, for soils such as those at Camp Lejeune. However, site-specific data were available for the calculation of K_d. Site-specific total organic carbon data (TOC) were available from samples collected between 1997 and 2007. These data can be used to calculate the soil-specific parameter, f_{oc}, representing the fraction of organic carbon in the soil or sediment. To calculate the K_d, f_{oc} is multiplied by the compound-specific parameter, K_{oc}.[184] K_{oc} is a constant representing the organic carbon partition coefficient, (i.e., the partitioning of a contaminant between organic carbon in the solid phase and the aqueous phase) and can be found in the literature. In fact, ATSDR listed compound-specific values for the contaminants considered in Camp Lejeune in their Table D12.[185] A tabulated list of the available data is provided in Appendix A.

The available data included some very low and very high values. To remove the potential effect of such very high or very low values on the calculation of the mean K_d, I calculated the geometric mean of the range instead. The geometric mean is less sensitive to extreme values, providing a measure that reflects the central tendency without being heavily influenced by very high or very low values. The geometric mean value for samples from all depths was 0.40 mL/g. When considering only samples collected near and below the water table (i.e., depths equal to or greater than 10 feet), the corresponding geometric mean was 0.42 mL/g. Both these values are greater than the value of 0.14 mL/g that ATSDR determined through model calibration.

Very high or very low K_d values can skew the calculated mean K_d value.[186] To further remove the impacts of very high or very low values in the dataset, I calculated the median value of the available data for samples from all depths, and from samples collected near and below the water table. The corresponding median values were 0.40 and 0.30 mL/g, respectively. These values are within the range that ATSDR considered from the literature for soils similar to those encountered in Camp Lejeune.

I calculated retardation factors corresponding to K_d values of 0.30 and 0.40 mL/g. For this calculation I used a bulk density of 1.65 g/cm^3 (which is appropriate for soils of the type found at Camp Lejeune and used by ATSDR later in their Hadnot Point model), and not the erroneous 2.7 g/cm^3 in ATSDR's calibrated model for Tarawa Terrace. I also used a porosity value of 20%, which is consistent with the value in the ATSDR model. The corresponding retardation factors for K_d values of 0.30 and 0.40 mL/g were 3.48 and 4.30, respectively, which are both greater than ATSDR's calibrated value of 2.93.

4.1.2.5 Model Estimated Monthly Contaminant Concentrations in the Influent to the WTP Would be Lower if the Source Starting Date and K_d Value Were Adjusted to Site-Specific Data

The resulting historical reconstruction of PCE concentrations at well TT-26 and the influent to the water treatment plant would be different if the following adjustments to the ATSDR model were made:

- Correct starting date for the ABC One-Hour Cleaners source (July 1, 1954); and

[184] Wiedemeier et al. (1999), p. 145

[185] ATSDR-TT, Chapter D, p. D15

[186] Helsel et al. (2020), p. 2

- K_d value equal to 0.3. This value corresponds to the median value of the site-specific data and is equal to the value ATSDR used in Hadnot Point. The corresponding retardation factor is 3.48.

The resulting historical reconstruction of PCE concentrations at well TT-26 and the influent to the WTP are depicted in Figure 7 and Figure 8, respectively. These graphical representations serve only to demonstrate how variable the model outputs are to changes in parameters. They should not be interpreted as definitive, mean monthly concentrations of contaminants in the influent to the WTP.

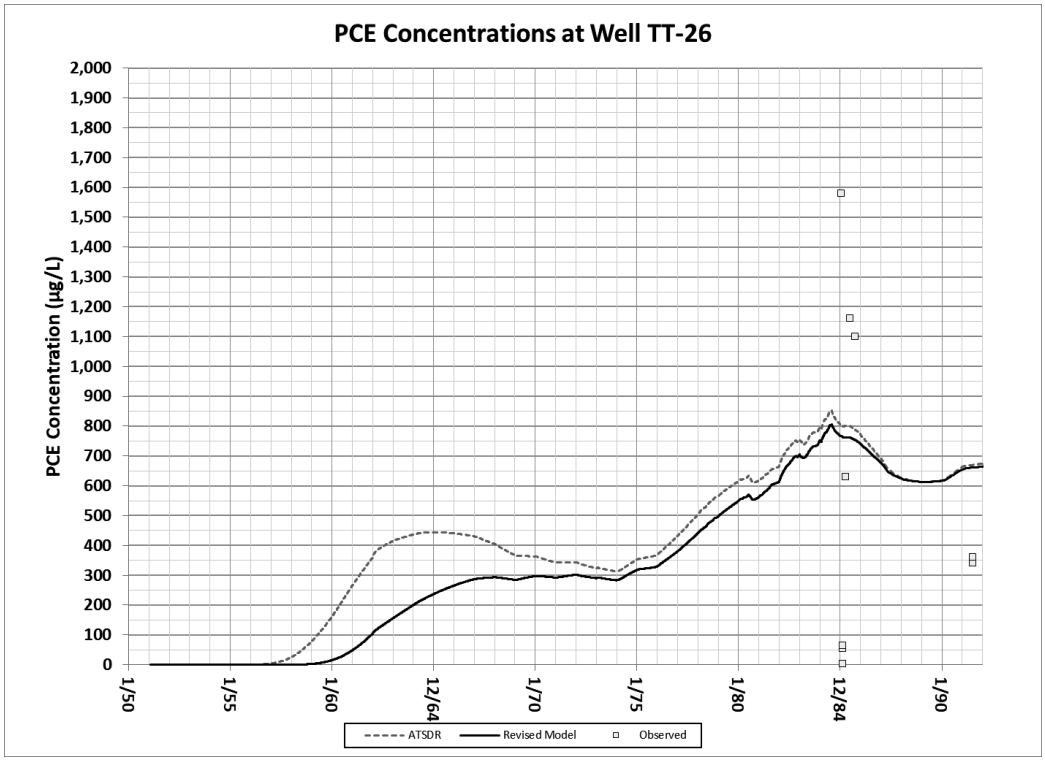


Figure 7: Model Simulated PCE Concentrations at Well TT-26 for Corrected Source Date and Retardation Factor 3.48

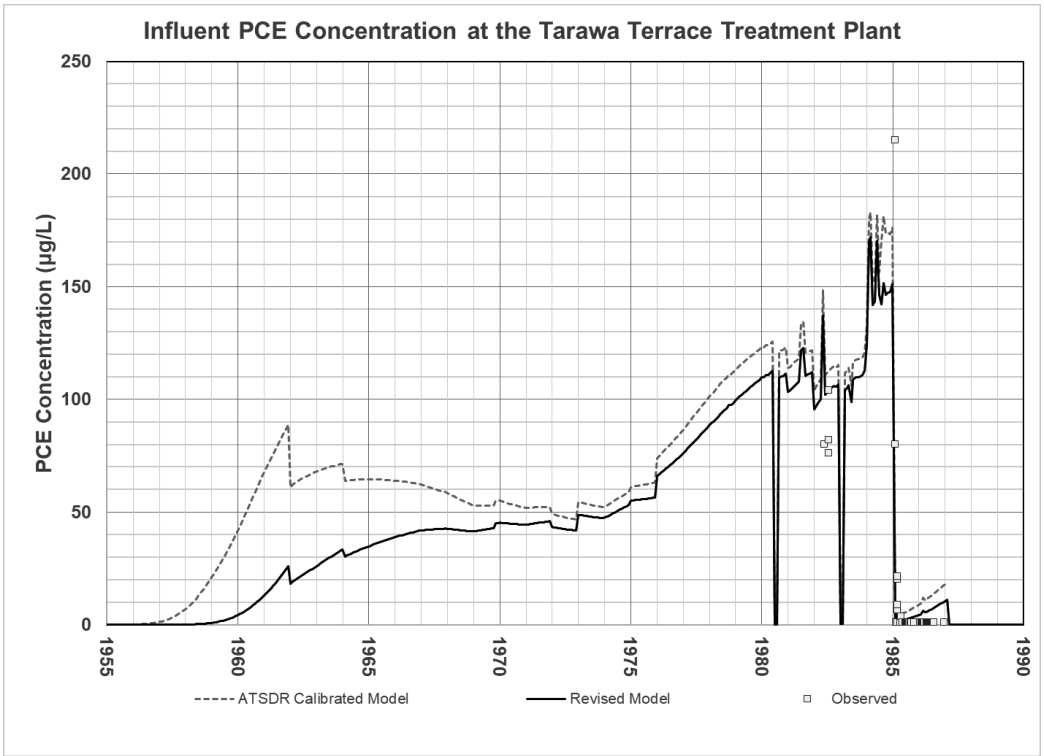


Figure 8: Model Simulated PCE Concentrations in the Influent to the TT WTP for Corrected Source Date and Retardation Factor 3.48

In Summary (Opinion 4): These model results illustrate how the ATSDR calibrated model, adjusted only to account for site-specific data for K_d, and correct implementation of the continuing source at ABC One-Hour Cleaners, estimates a historical reconstruction that fits the data equally well. It also indicates (a) a slower arrival of contamination to well TT-26 and, therefore, the influent to the WTP, and (b) much lower concentrations than those calculated by ATSDR over a period of about 15 years from the initial PCE releases. The uncertainty range for such historical reconstruction would also be lower, as it would be based on slower plume migration and lower concentrations for many years after the start of contaminant releases from the source.

This also demonstrates that using a groundwater model for hindcasting is highly uncertain in the absence of site-specific data for assigning parameter values and a lack of observed data to constrain the model calibration. While professional judgment is essential in model construction, it cannot guarantee model accuracy absent these data.

4.1.2.6 Elevated Concentrations at Pumping Well TT-23 Are Biased High and Inflate WTP Concentrations

The ATSDR model calculated monthly concentrations at each well. ATSDR argued that those concentrations were within the calibration standards. However, concentrations at TT-23 were much higher, almost two times as high as the measured concentrations. In addition, following the well being shut down, measured concentrations dropped precipitously to non-detections, but ATSDR model's simulated concentrations remained elevated.

ATSDR explained this discrepancy by suggesting different factors for the elevated model-simulated concentrations.[187] It is not possible to test ASTDR's hypotheses for explaining these enormous differences. ATSDR contended that such factors are not built into the model, and therefore, the simulated concentrations would inevitably be higher than the measured ones.

Nonetheless, ATSDR did not apply any adjustment to the simulated concentrations to reflect the admittedly lower concentrations at TT-23. Instead, ATSDR used those high model-simulated concentrations in their mixing model, thereby inflating the calculated PCE concentrations in the influent to the WTP.

In Summary (Opinion 5): ATSDR's model simulated substantially higher concentrations at supply well TT-23 than indicated by observed data. Although ATSDR admitted the discrepancy, they did not correct these elevated concentrations before using them for calculating the mixed concentration in the influent to the WTP. As a result, they inflated the estimated monthly contaminant concentrations at the WTP for several months during which well TT-23 was in service.

4.1.2.7 ATSDR's Model Calibration Did Not Fully Consider Non-Detections and Overestimated Plume Migration

Comparing the observed versus simulated concentrations highlights additional issues regarding model performance. ATSDR included non-detections in its model calibration, but the calibrated model did not reflect those observed non-detections.

ATSDR constructed a scatterplot of simulated and observed PCE concentrations.[188] Non-detections were not shown in that figure, as non-detections listed as zeros are not visible in a logarithmic-scale scatterplot. This is because a logarithmic scale can only show numbers greater than zero. I replaced the observed non-detections with a value of 0.1 and I reconstructed the scatterplot so that observed non-detections are not hidden from the plot. Figure 9 shows both scatterplots for comparison.

In the plot on the right in Figure 9, the points along the vertical axis indicate non-detections in field samples corresponding to higher concentrations calculated by the model, encompassing a range that extends to hundreds of micrograms per liter (μg/L). Most of these calculated concentrations are within ATSDR's arbitrary range of acceptable calibration results. But, this reveals some important issues with the calculated concentration trends.

[187] ATSDR-TT, Chapter A, p. A25; ATSDR-TT, Chapter F, p. F32

[188] ATSDR-TT, Chapter F, Figure F12, p. F33

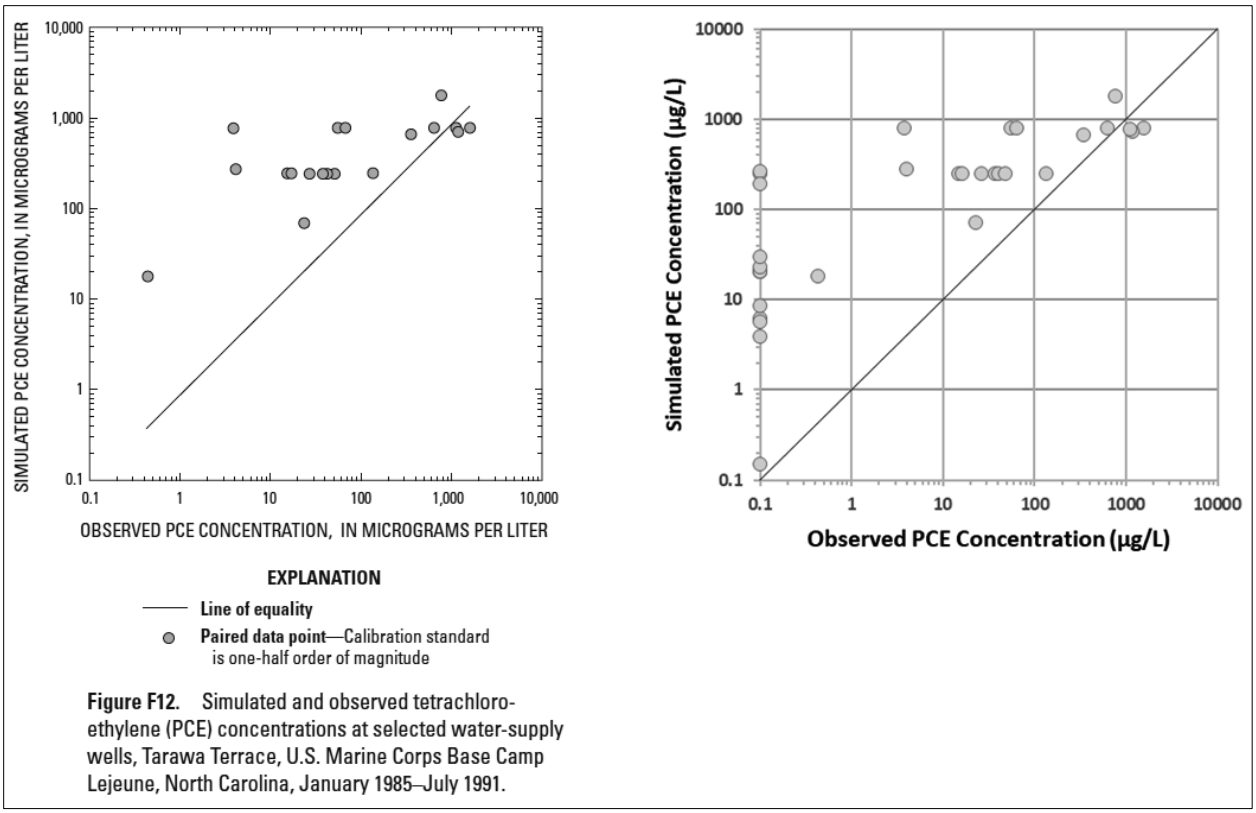


Figure F12. Simulated and observed tetrachloroethylene (PCE) concentrations at selected water-supply wells, Tarawa Terrace, U.S. Marine Corps Base Camp Lejeune, North Carolina, January 1985–July 1991.

Figure 9: Comparison of Calibration Scatterplots With and Without Non-Detections

Figure 10 shows the plots produced by ATSDR, illustrating the historical reconstruction of concentrations at the pumping wells based on the calibrated model results, against the available observed concentrations.[189] Brown arrows have been added to those plots to indicate periods when simulated concentrations trend upward, contrary to what the observed data indicate. Also, red arrows have been added to indicate the significant discrepancies between simulated and observed data.

According to plot F15 for well TT-25, model results indicate that PCE arrived at that well around 1983 and concentrations continued increasing. It should be noted that this plot only shows the reported J-value[190] of 0.48 µg/L in September 1985, and not the additional observed data listed in ATSDR's Table F13, which indicate non-detections from February to December 1985. It should also be noted that the horizontal axis in this plot is not at a scale appropriate for clearly identifying the dates of the individual samples.[191]

[189] Plots of simulated and observed PCE concentrations in wells (ATSDR-TT, Chapter F, pp. F34 and F35); PCE plume map in model Layer 3 in December 1984 (ATSDR-TT, Chapter F, p. F40).

[190] J values represent a reported concentration below the detection limit of the instrument for the implemented method of analysis in the laboratory, but with sufficient "noise" to be estimated at a very low level.

[191] It appears that the single observed data point is not plotted at the correct date, as the model-simulated concentration at that date is much higher than the value of 0.48J of September 25, 1985, reported in ATSDR's Table F13 (ATSDR-TT, Chapter F, p. F33).

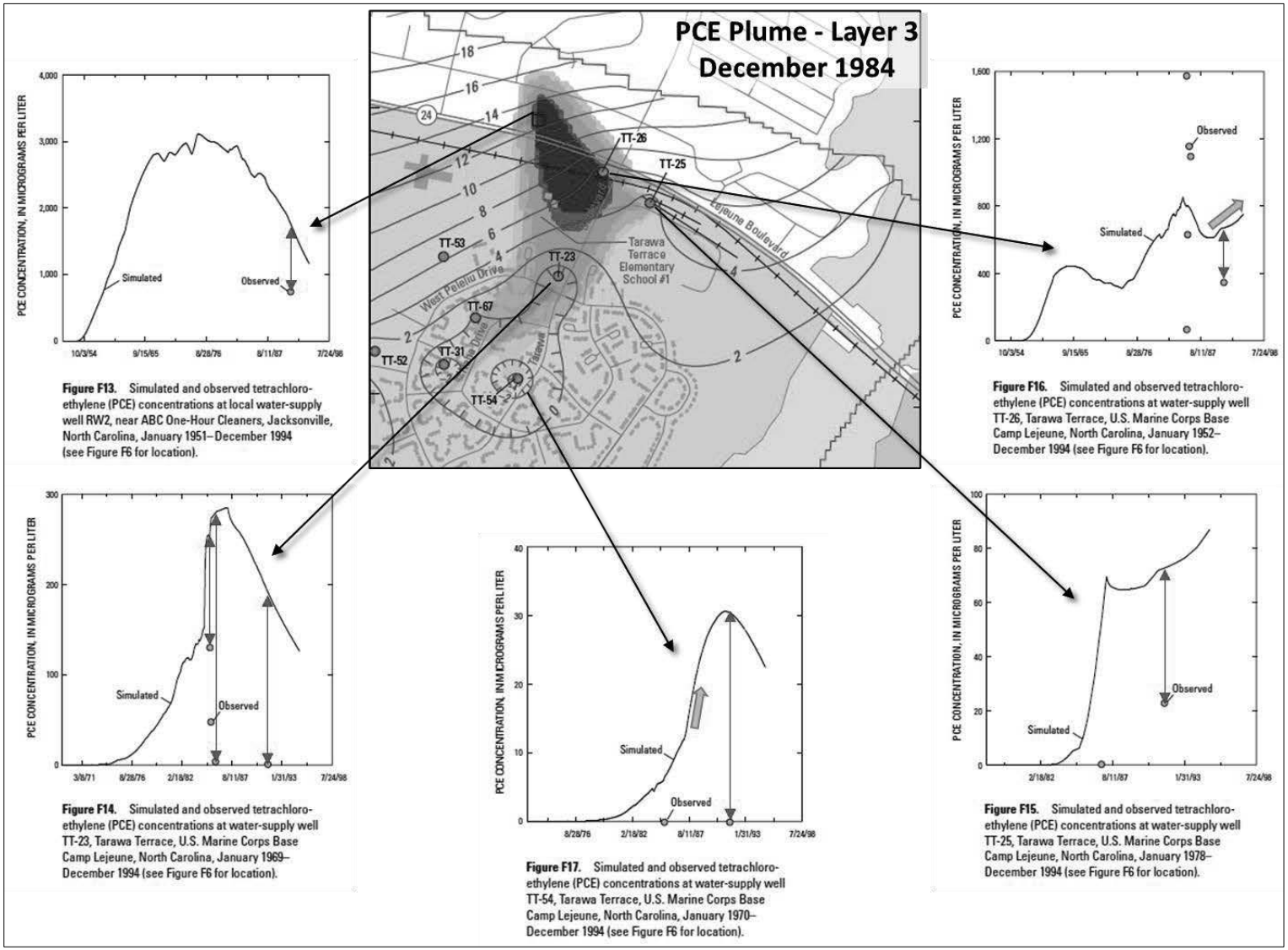


Figure F13. Simulated and observed tetrachloroethylene (PCE) concentrations at local water-supply well RW2, near ABC One-Hour Cleaners, Jacksonville, North Carolina, January 1951–December 1994 (see Figure F6 for location).

Figure F16. Simulated and observed tetrachloroethylene (PCE) concentrations at water-supply well TT-26, Tarawa Terrace, U.S. Marine Corps Base Camp Lejeune, North Carolina, January 1952–December 1994 (see Figure F6 for location).

Figure F14. Simulated and observed tetrachloroethylene (PCE) concentrations at water-supply well TT-23, Tarawa Terrace, U.S. Marine Corps Base Camp Lejeune, North Carolina, January 1969–December 1994 (see Figure F6 for location).

Figure F17. Simulated and observed tetrachloroethylene (PCE) concentrations at water-supply well TT-54, Tarawa Terrace, U.S. Marine Corps Base Camp Lejeune, North Carolina, January 1970–December 1994 (see Figure F6 for location).

Figure F15. Simulated and observed tetrachloroethylene (PCE) concentrations at water-supply well TT-25, Tarawa Terrace, U.S. Marine Corps Base Camp Lejeune, North Carolina, January 1978–December 1994 (see Figure F6 for location).

Figure 10: Comparison of Observed and ATSDR Model-Calculated Concentrations at Pumping Wells

The model results also indicate a low value of 5.8 µg/L at the distant pumping well TT-54, south of well TT-23 in February 1985, when the observed value is non-detection. In fact, plot F17 indicates that the calibrated model estimated the first arrival of the PCE plume at that well around 1978, with concentrations increasing after that time. This is not supported by the non-detection in the sample collected in February 1985.

After 1985, when the wells were shut down, concentrations simulated by the ATSDR model continued to rise. However, reported concentrations from groundwater samples taken at wells TT-26, TT-25 and TT-23 indicated decreasing concentrations, compared to the period when the wells were operating (plots F15 and F16 in Figure 10).

The discrepancies between observed and simulated concentrations at well TT-23 (Section 4.1.2.6) are illustrated in plot F14 in Figure 10.

Well TT-54 had a reported non-detection in July 1991. However, the ATSDR model indicated an increasing concentration trend at well TT-54, suggesting that the PCE plume continued arriving at that well until that time.

This is unlikely to be accurate. ATSDR's Tarawa Terrace model results overestimated observed concentrations, the extent of the contaminant plume, and simulated concentrations after the pumping wells were turned off.

In Summary (Opinion 6): The model was not reliably calibrated. Model results indicate biased-high estimates of contaminant mass in the aquifer, where observed data indicate the absence of contamination. Simulated concentrations at well TT-26 andwell TT-54, located thousands of feet south of well TT-26, trended upward when observed data indicated a downward trend or no contamination, respectively. ATSDR's model overestimated the plume migration extent and rate of migration, which were both conservative and biased-high.

4.1.3 ATSDR's Uncertainty Analysis was Limited and Biased-High

ATSDR states that the Tarawa Terrace groundwater model is "*subject to varying degrees of uncertainty which are associated with: (1) limited or lack of data, (2) erroneous data due to precision and accuracy limitations, and (3) simplifications of mathematical equations represented by the model*."[192] However, according to ATSDR, "*[t]hese probabilistic results provide additional confidence that the deterministically derived results (for example, the historically reconstructed PCE concentrations in Tarawa Terrace finished water) are reasonable and conform well to field observations and data."*[193]

Regarding model calibration and uncertainty analysis, "*realizations are all constrained by the necessity to respect expert knowledge and the need to allow the model to replicate observed system behavior*."[194] It is therefore important for the model to "replicate observed system behavior." Here, observed system behavior refers to the measured or observed data taken from Tarawa Terrace water supply wells and the WTP. Recall the discussion on model precision and accuracy.

The uncertainty analysis of ATSDR's Tarawa Terrace model did not consider the "observed system behavior," as historical data were not available to guide this analysis. Recall (Section 4.1.2) that Dr. Clement expressed concerns about the fact that the ATSDR "*model was calibrated to limited number of*

[192] ATSDR-TT, Chapter I, p. I3

[193] ATSDR-TT, Chapter I, p. I54

[194] Sepulveda and Doherty (2015)

datapoints,"[195] which Maslia et al. (2012) did not address in their response. As a result, the calibrated model and uncertainty analysis focused on model precision and not accuracy. Recall Figure 2 in Section 3.1.5: ATSDR's calibration and uncertainty analyses results are precise but not accurate, as described by the bottom-left bullseye in the graphic.

As will be discussed in the following sections, ATSDR's uncertainty analysis relied solely on the parameters and results of the calibrated model for estimating its uncertainty range, on the premise that the calibrated model successfully reconstructed the history of contamination at the site. The significant implications of the lack of historical data and assumptions made by ATSDR and implemented in the model are discussed below.

4.1.3.1 ATSDR's Presentation of Uncertainty Analysis Results is Visually Misleading

ATSDR stated that the uncertainty analysis they conducted provides confidence that model results are reasonable and conform to observed data. However, ATSDR's presentation of its uncertainty analysis results is visually misleading because they used a logarithmic scale, which visually compresses the uncertainty range around their calibrated model. In Figure 11, the uncertainty range appears to be a narrow band enveloping the calibrated model. However, the logarithmic scale for PCE concentrations on the vertical axis of Figure 11 spans over six orders of magnitude.[196]

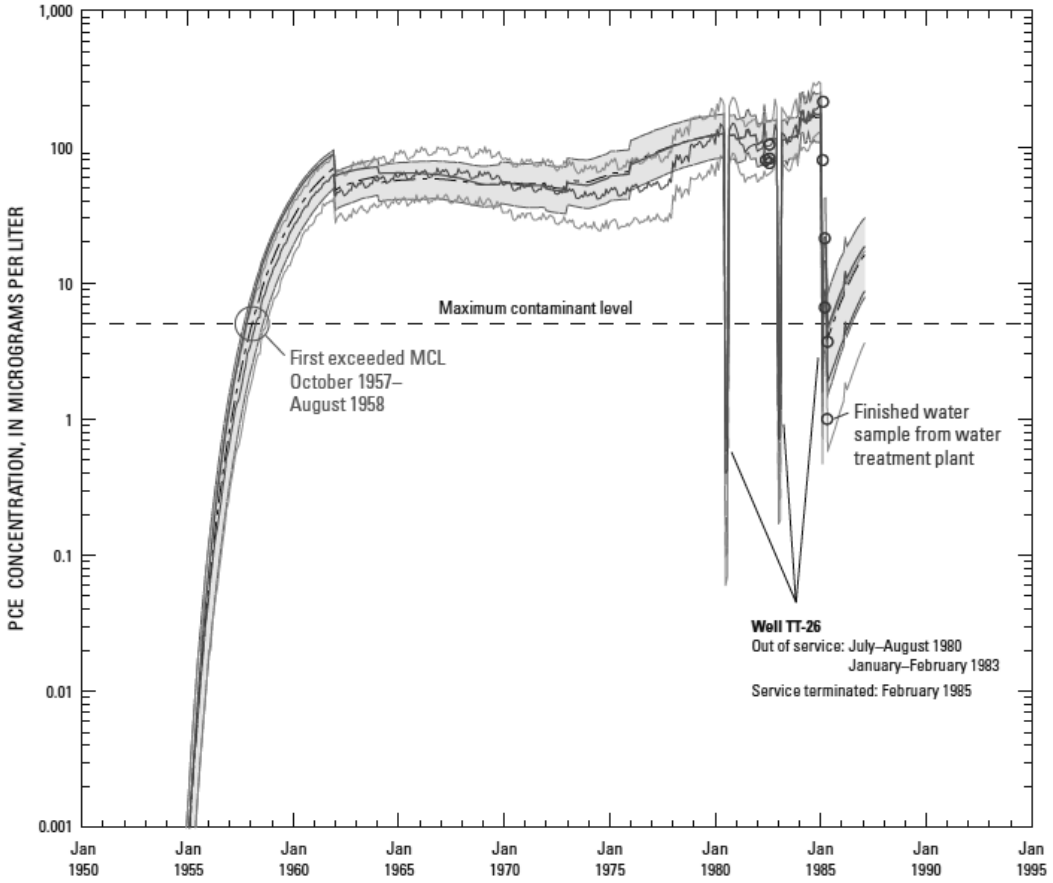
Logarithmic scales are appropriately used to visualize a wide range of values, where data span many orders of magnitude. This makes it easier to visualize and compare results. However, the difference between the high and low values in Figure 11 is not significant enough to justify the use of a logarithmic scale. The reconstructed historical concentrations only vary between non-detections and about 200 µg/L, i.e., a range of approximately two orders of magnitude, and not the six orders of magnitude displayed on Figure 11. By displaying a wide range of simulated PCE concentrations increasing exponentially, Figure 11 visually condenses the range of uncertainty around the calibrated model. This is not appropriate for the data presented in the figure, because the extent of the uncertainty range and the performance of the calibrated model within that range are not easily visible.

For these data, an arithmetic scale would be more appropriate to illustrate the extent of uncertainty range in a visually-accurate manner. A modified version of Figure 11 using an arithmetic scale for PCE concentrations on the Y-axis is provided in Figure 12.

Figure 12 depicts the uncertainty range calculated by ATSDR for two scenarios: (a) without considering pumping uncertainty (yellow shaded area), and (b) including pumping uncertainty (red lines). Also shown in this figure is the historical reconstruction of PCE concentrations in the influent to the WTP, simulated by ATSDR's calibrated model (black line).

[195] Clement (2011)

[196] An order of magnitude is a range of magnitude extending from some value to ten times that value (Merriam-Webster Dictionary). For example, the values 12 and 120 are separated by an order of magnitude. More generally, the values 12 and 253 can be said that they are separated by about an order of magnitude.



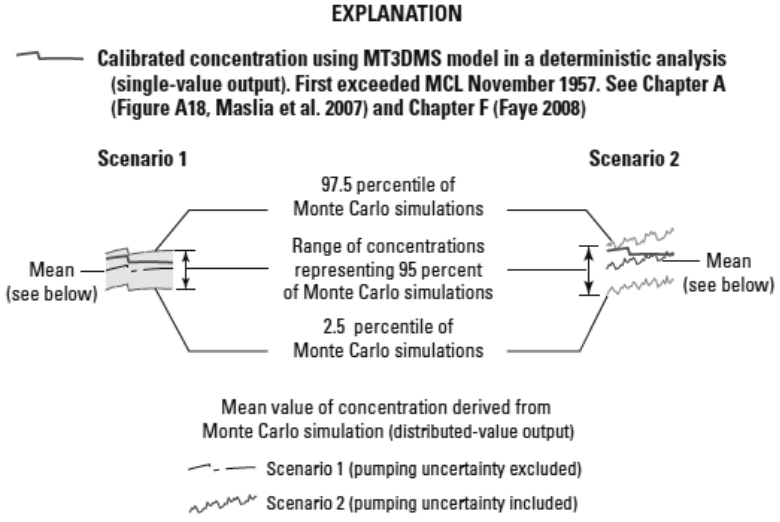


Figure I29. Concentrations of tetrachloroethylene in finished water at the water treatment plant derived from scenario 1 (pumping uncertainty excluded) and scenario 2 (pumping uncertainty included) probabilistic analyses using Monte Carlo simulation, Tarawa Terrace, U.S. Marine Corps Base Camp Lejeune, North Carolina.
[See Appendix I5 for tabular listing; PCE, tetrachloroethylene; MCL, maximum contaminant level]

Figure 11: Copy of Figure I29 from ATSDR's Probabilistic Analysis

As shown in Figure 12, concentrations calculated by the calibrated model are at the upper bound of the uncertainty range in the early years. In a rigorously conducted uncertainty analysis, the concentrations calculated by the calibrated model should be generally in the middle of the uncertainty range (Section 3.1.5). However, the calibrated model-simulated concentrations are almost identical to the upper bound of the uncertainty range in the early years of operation (1957-1963). This demonstrates that the simulated arrival times of PCE at well TT-26 and, as a result, at the WTP, are biased early. Plume migration in the calibrated model is biased high due to the retardation factor selected by ATSDR.

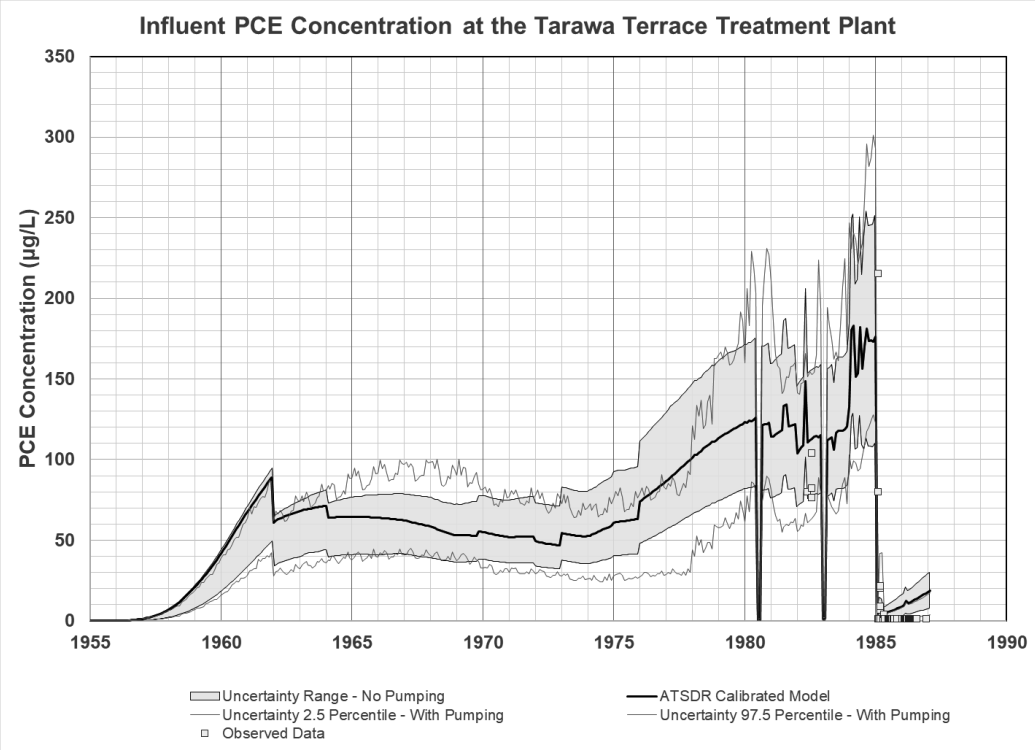


Figure 12: Reconstruction of ATSDR's Figure I29 - Probabilistic Analysis in Linear Scale

In Summary (Opinion 7): ATSDR presented the results of the uncertainty analysis using a format that was visually misleading. The choice of a logarithmic scale ranging over six orders of magnitude depicted a narrow uncertainty range around the calibrated model. However, using a linear scale for the same graph, the results indicate that the uncertainty range is broad and the estimated monthly concentrations are conservative and biased-high in the early years.

4.1.3.2 ATSDR's Parameter Ranges in the Uncertainty Analysis are Narrow and Biased

As discussed above in Section 3.1.5, an uncertainty analysis is designed to provide a range of possible model outcomes. A Monte Carlo uncertainty analysis, such as the analysis implemented by ATSDR, is a probabilistic uncertainty analysis. Its goal is to quantify the range and likelihood of model outputs.[197] This is accomplished by running the model many times. Each run is referred to as a realization. Each time the model is run, different permutations of parameter values are selected from the range of

[197] ATSDR-TT, Chapter I, p. I30

possible values for each parameter. The modeler selects these parameter value ranges based on the modeler's professional judgment and consideration of field conditions or a literature review.

ATSDR conducted the uncertainty analysis considering a total of 840 realizations.[198] Parameter values varied within a defined range, and values from each range were selected for each realization. The uncertainty analysis was conducted in stages. First, the groundwater flow model was run for each realization, and "*physically implausible realizations*" were discarded using criteria for model fit to data.[199] By doing so, ATSDR acknowledged the importance of generating realizations that would "calibrate" the model and replicate observed conditions.

Then, the contaminant fate and transport model was run for the remaining, physically plausible realizations.[200] ATSDR could not identify and discard similarly "implausible realizations" for the fate and transport model, because there were no concentration data available during the historical period prior to 1982. This means that the complex process implemented for the uncertainty analysis would result in precise but not necessarily accurate solutions, as the latter could not be validated based on observed data.

ATSDR's uncertainty analysis considered multiple permutations of model parameters. I will focus on one, the retardation factor. The variability in retardation factors was limited, impacting the range of uncertainty estimates.

Before discussing the specifics of the implementation of ATSDR's uncertainty analysis, an example is provided to illustrate conceptually how this analysis was conducted and why it was not expansive enough to account for retardation factor variability at Tarawa Terrace.

Assume that 100 cars travel from Point A to Point B, all departing at the same time with little to no traffic. Given the posted speed limits, average speeds will not vary much. Hence, all cars will cover the distance, 200 miles, at the same or about the same travel time, 4 hours, and, travel at the same or about the same average speed, say 50 miles per hour. Does that mean we can conclude, with a high degree of certainty, it always takes 4 hours to go from Point A to Point B?

The answer is no. For example, what if the cars departed during rush hour, rather than when there was no traffic? Then, travel times would be longer.

In order to be sure that the range of travel times is reasonably estimated, additional factors like traffic must be considered. By doing so, the range of calculated travel times would certainly be wider, including much longer travel times.

The importance of this concept is illustrated when examining how ATSDR performed an uncertainty analysis to evaluate how fast PCE would arrive at well TT-26 and at what concentrations. In this analogy, the cars are PCE, the highway between Points A and B is the groundwater in the aquifer, and traffic is the retardation factor.

One of the most critical parameters for determining how fast contaminants will migrate in the aquifer is the retardation factor, which is calculated using K_d, bulk density, and porosity. ATSDR evaluated the effects of retardation in their uncertainty analysis by doing something similar to the 100-car example.

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[198] Two Monte Carlo uncertainty scenarios were considered: Scenario 1, excluding pumping uncertainty, and Scenario 2, including pumping uncertainty. Stopping criteria were applied to determine whether a realization was successfully completed, or it should he halted. Of the 840 realizations, 510 realizations were successfully completed for Scenario 1, and 684 realizations were successfully completed for Scenario 2. See ATSDR-TT, Chapter I, p. I48.

[199] ATSDR-TT, Chapter I, p. I31

[200] ATSDR-TT, Chapter I, p. I31

ATSDR developed hundreds of scenarios (realizations) where, theoretically, retardation factors would vary within some range, in order to assess the impact on travel time of contamination.

To achieve that, ATSDR first defined ranges of values for K_d, bulk density, and porosity. Every cell in the model was assigned a random value from the range of each parameter. Therefore, every cell had a different retardation factor. ATSDR repeated this process to develop model inputs for 840 realizations. When inspecting the inputs of these realizations, I observed the following:

The model calibrated K_d was 0.14 mL/g, and the assigned uncertainty range, as implemented in the Monte Carlo realizations, was between 0.11 and 0.31 mL/g. Although this range suggests that the higher K_d values from the site-specific data are within this range and, therefore, they were considered by ATSDR in their uncertainty analysis, this was not the case. This is because ATSDR implemented a "probability distribution function," which is a term to describe how values closer to the mean value of the range are more probable than those away from the mean. In ATSDR's uncertainty analysis, the defined "probability distribution function" resulted in selecting almost 85% of the values from a fraction of the range, between 0.11 and 0.20 mL/g.

Recall that a lower K_d value is associated with lower retardation factors, and therefore faster movement of contaminants through the subsurface. By using mostly smaller values, ATSDR's selection process tended towards the equivalent of a light to no traffic scenario in the 100-car example, and ignored the possibility of rush-hour traffic. Hence, ATSDR leaned toward faster plume migration, resulting in earlier arrival of contamination at well TT-26.

ATSDR followed a similar process to define ranges and select values for the bulk density and porosity.

I further deconstructed the calculation of the uncertainty range and focused on the variability of retardation rates. I simulated influent concentrations to the WTP by running the model 840 times, using ATSDR's retardation inputs developed for their uncertainty analysis as described above, and keeping all other inputs in the model unchanged. The results of these calculations are depicted in Figure 13.

In Figure 13, ATSDR's calibrated reconstruction of historical concentrations is shown by the black line. The range of historical concentrations due to the variability of the retardation range is shown by the blue shaded area, calculated from my 840 realizations. The uncertainty range for the non-pumping scenario is shown by the yellow shaded area. This figure illustrates that the uncertainty due to the variability in retardation factors is very small relative to the overall uncertainty range calculated by ATSDR. This is because ATSDR's range of retardation factors was very narrow.

Figure 13 also illustrates that the calibrated model sits at the upper bound of the retardation-factor uncertainty range (black line at the top of the blue shaded area). This is also an indication of bias, as the calibrated model should be generally in the middle of the uncertainty range (Section 3.1.5). ATSDR's selection of the retardation-factor parameters forced the calibrated model to simulate the fastest arrival of PCE at well TT-26 and, from there, the treatment plant. ATSDR's uncertainty analysis indicated that plume migration could only be slower and at lower concentrations in the early years, when retardation factors are considered.

To further investigate why this happened, I calculated average retardation factors from layers 1 and 3 of the model for each realization and the range of those averages over the 840 realizations. Layer 1 of the model represents the unconfined layer directly beneath the earth's surface, where the contaminant mass was introduced to the aquifer from the contaminant source. Layer 3 of the model represents the aquifer below the confining unit, where water supply well TT-26 was screened. For this investigation, I evaluated how retardation factors varied on average between the contaminant source and the supply well. My evaluation concluded that average retardation factors only varied over a very narrow range in ATSDR's uncertainty analysis. To understand the importance of this, recall the 100-car example.

In that example, each car may travel at different speeds over different sections of the highway. Assuming these sections are of equal length, such as the cells of the model, the total travel time for each car is determined by its average speed over all the sections of the highway. If cars traveled at variable speeds over different sections, but their corresponding average speed over the entire distance was about the same, the total travel time would be about the same for all cars. Each car could drive faster or slower over different portions of the highway, but if their speeds averaged out to the same value, the travel times also average out to the same value. Hence, in the case of retardation factors and travel times, the small variability of the retardation factors means the overall variability of contaminant travel times is also small.

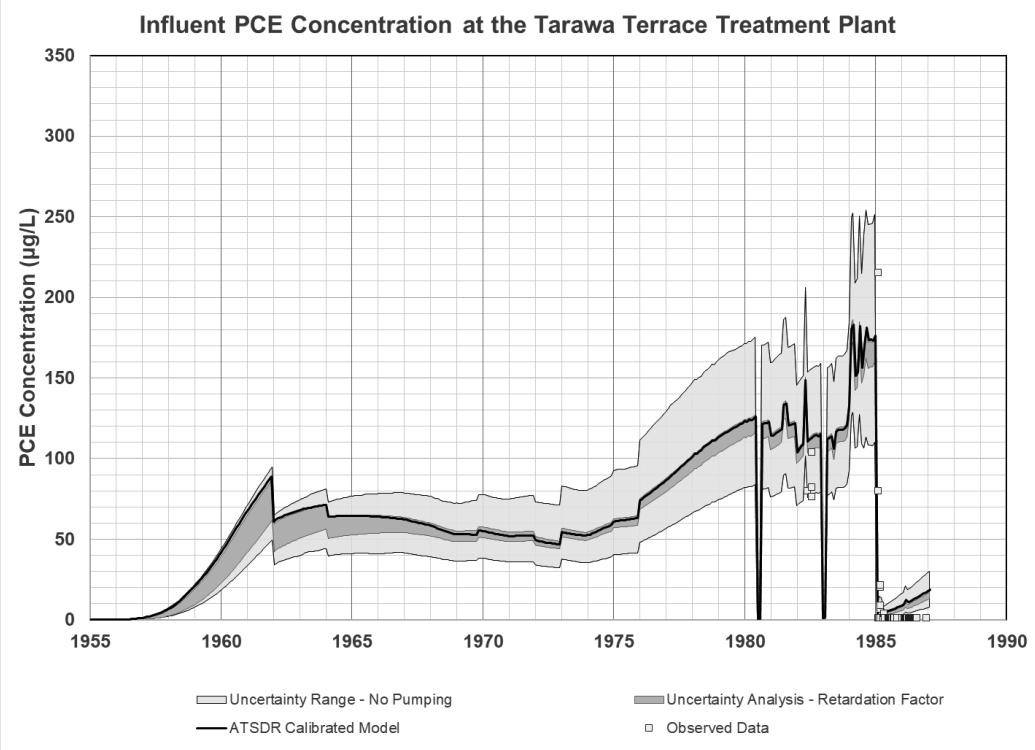

Figure 13: PCE Concentrations from Calibrated Model and Retardation-Factor Uncertainty Bounds

This means that the uncertainty analysis did not evaluate a larger portion of the range of possible retardation factors based on the parameter ranges ATSDR defined as reasonable for the site. By doing so, ATSDR ignored the possibility of slower plume migration in the aquifer and later arrival times of contamination at the water supply well. This is not consistent with a rigorous and appropriate uncertainty analysis.

My evaluation concluded that, in Layer 1, the average retardation factor over the 840 realizations ranged from 3.275 to 3.296. In Layer 3, the average retardation factor over the 840 realizations ranged from 3.276 to 3.297. Repeating the same steps above to calculate the median values over the 840 realizations in Layers 1 and 3, I determined that the median values in Layer 1 ranged from 3.122 to 3.146. I also determined that the median values in Layer 3 ranged from 3.123 to 3.148.[201]

This means that, despite the range of parameter values assigned to model cells, mean and median parameter values varied very little. This is not entirely surprising, as the probability distribution functions are constructed to assign parameter variability around the calibrated value of the parameters. But it also

[201] See Appendix C for details

illustrates the inherent limitation of the ATSDR's uncertainty analysis, which is not based on site-specific data for the parameter values and is not informed by historical observed data. Instead, reconstructed historical concentrations vary over a very narrow range selected by ATSDR, as illustrated in Figure 13.

If a wider range of retardation factors were considered, simulated concentrations at the pumping wells would have been different. As an example, Figure 14 illustrates the historical reconstruction of PCE concentrations at pumping well TT-26, comparing the ATSDR calibrated model to an alternative version that uses a retardation factor of 4.3.[202] In this alternative version, the retardation factor of 4.3 is higher than ATSDR's calibrated value of 2.93 and outside the range of average and median retardation factors considered in their uncertainty analysis (3.122 to 3.297, see above). The comparison between the two versions indicates that it is possible to have a different reconstruction than ATSDR's, but that would still have fit within ATSDR's definition of a calibrated model. This is based on ATSDR's calibration criteria of ± one-half order of magnitude and target-range violations that ATSDR considered acceptable for their calibrated model. This alternative reconstruction also lies within ATSDR's calculated uncertainty range during the period when data are available. ATSDR did not consider a wider range of retardation values because they constrained the ranges of key parameter values, such as K_d, to low levels.

Figure 15 depicts a comparison of influent concentrations at the TT WTP, calculated by the ATSDR calibrated model, and my alternative version using the higher retardation factor. This figure illustrates that the historical reconstruction calculated using a higher retardation factor is comparable to ATSDR's but outside the uncertainty range calculated for retardation factors based on the 840 realizations for retardation factors that ATSDR developed.

Figure 16 presents a similar comparison, where the results of the ATSDR model and those of its alternative version with the higher retardation factor are shown. In this figure, additional lines are included, showing the total uncertainty bounds that ATSDR calculated, which included the collective uncertainty of all parameters, including pumping uncertainty (i.e., uncertainty evaluation with and without varying the historical pumping configuration). This figure shows that:

- Timeframes of PCE arrival at the WTP could be longer than those estimated by ATSDR, both in its calibrated model and its uncertainty analysis; and
- PCE concentrations at the WTP would be lower than those calculated by ATSDR for at least 10 years, when considering either its calibrated model or even its complete uncertainty analysis considering all parameter uncertainties.

If ATSDR considered higher retardation rates, it could have developed a different calibrated model that would be equally plausible and consistent with site-specific data. This alternative model would then provide a new basis for evaluating uncertainty, as its calibrated parameters would be used for determining corresponding parameter ranges – which, in the case of the retardation factor, the parameter range would have been shifted to higher values than those considered by ATSDR. As a result, the uncertainty range would have been possibly wider, and its bounds would be lower than those calculated by ATSDR.

The impact of using a more appropriate parameter range would be further exacerbated if the correct starting date for mass loading at the ABC One-Hour Cleaners source was used. With those corrections, PCE arrival times at well TT-26 and, therefore, in the influent to the WTP would be longer than those estimated by ATSDR.

ATSDR selected a range of acceptable values for key parameters, such as K_d, for their uncertainty analysis based solely on professional judgment and literature sources. However, in their analysis, they considered a smaller subset of that range. In addition, the average values of those parameters in their

[202] This retardation factor is based on a K_d value of 0.40 mL/g from site-specific data (see Section 4.1.2.4) and a bulk density of 1.65 g/cm^3.

realizations varied only slightly. As a result, the uncertainty range of the simulated reconstructions for those parameters was narrow and biased high.

When I modified the parameter values in the model to be within the range of site-specific data, the simulated reconstruction was outside ATSDR's uncertainty range. This means that ATSDR's model calibration did not consider appropriate parameter values based on site-specific data. Also, ATSDR's uncertainty analysis was not inclusive of the entire range of parameter values. This demonstrates that the results of model calibration and uncertainty analysis are unreliable in the absence of site-specific data for parameter assignment and a lack of observed data to constrain the model calibration.

ATSDR's upper and lower bounds of uncertainty, shown in Figure 13, that represent 95% of Monte Carlo simulations[203] are therefore unreliable, conservative, and biased-high based on the discussion above.

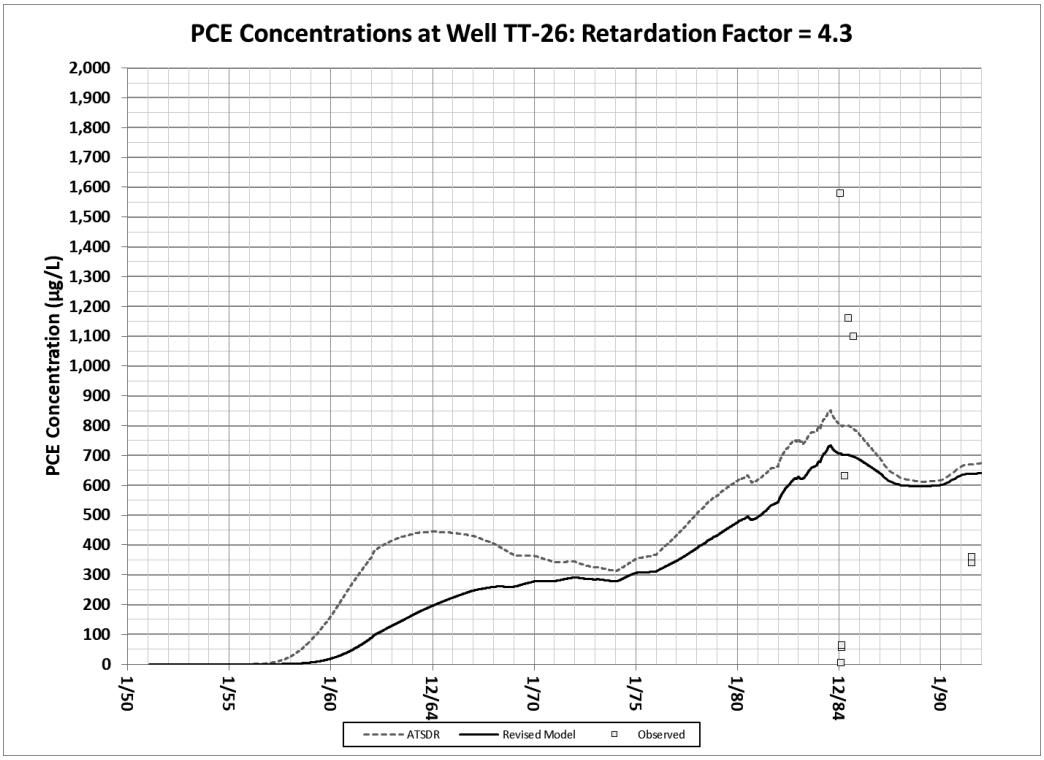

Figure 14: PCE Concentrations at Well TT-26 for Retardation Factor 4.3

[203] 2.5 and 97.5 percentile.

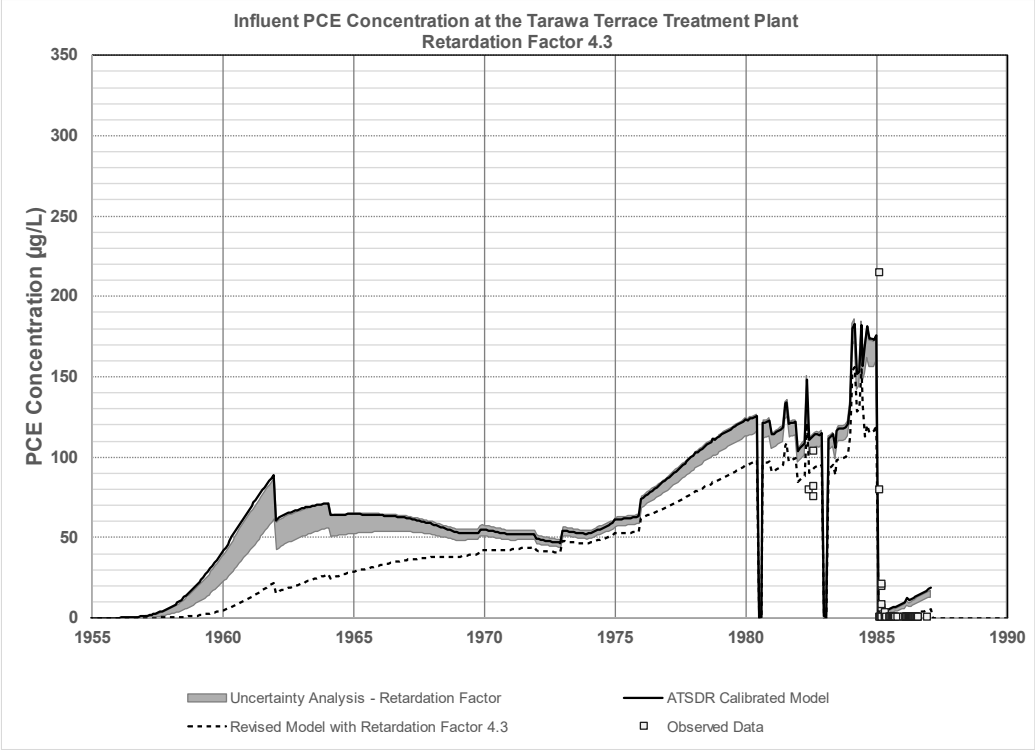

Figure 15: Influent PCE Concentrations at the Tarawa Terrace Water Treatment Plant for Retardation Factor 4.3

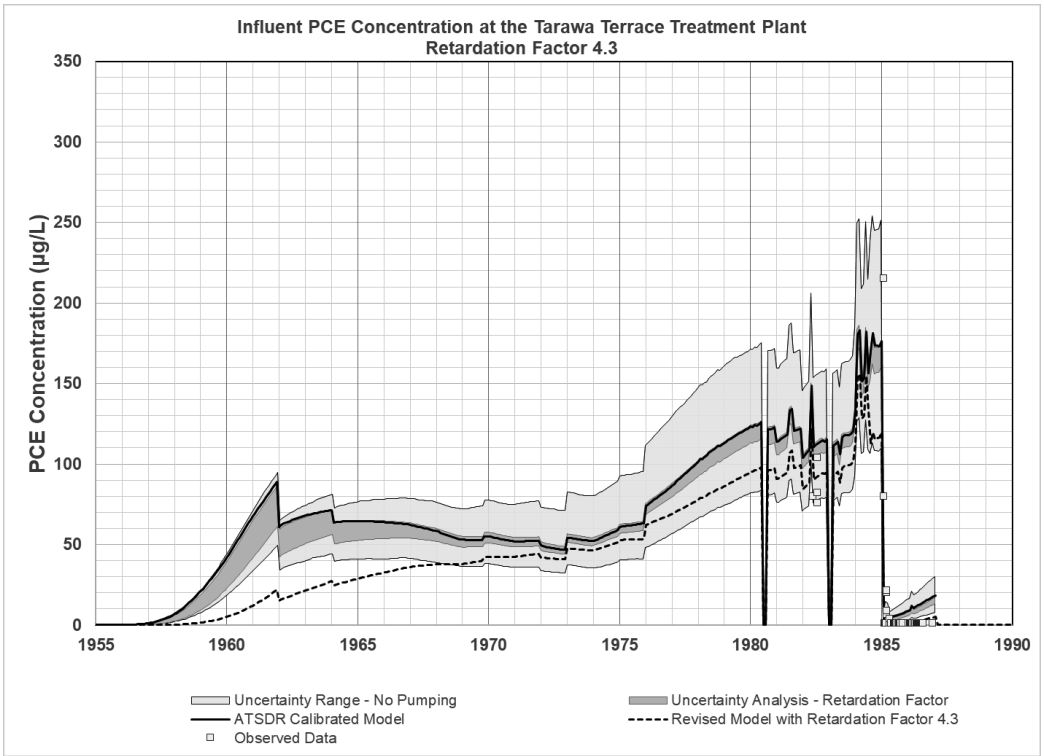

Figure 16: Influent PCE Concentrations at the Tarawa Terrace Water Treatment Plant for Retardation Factor 4.3: ATSDR vs. Higher Retardation Model

In Summary (Opinions 8 and 9): The results of ATSDR's model calibration and uncertainty analysis are unreliable in the absence of site-specific data for parameter assignment and a lack of observed data for constraining the model calibration. The uncertainty analysis was not bound by historical concentration data, and as a result, focused only on model precision and not accuracy in predicting COC concentrations.

ATSDR's uncertainty analysis did not evaluate a wider range of possible retardation factors, even based on the parameter ranges ATSDR considered reasonable for the site. Hence, ATSDR ignored the possibility of slower plume migration in the aquifer and later arrival times of contamination at the water supply well. This is not consistent with a rigorous and appropriate uncertainty analysis.

ATSDR's selection of the retardation factor parameters forced the calibrated model to simulate the fastest arrival of PCE at well TT-26 and, from there, the treatment plant. ATSDR's calibrated model was biased-high, and the uncertainty analysis indicated that plume migration could only be slower and at lower concentrations in the early years, when retardation factors are considered. If ATSDR considered higher retardation rates, within the range of site-specific data, simulated plume migration would be slower, and reconstruction estimates of monthly contaminant concentrations would be outside ATSDR's uncertainty range.

4.1.4 VOC Degradation By-Products: MT3DMS vs. TechFlowMP Model Results for PCE are Inconsistent and Biased High, and Should be Corrected for Site-Specific K_d Values

ATSDR developed and calibrated groundwater flow and contaminant transport models, using MODFLOW and MT3DMS, to reconstruct historical concentrations of PCE in groundwater extracted by the pumping wells, and the mixed water from the pumping wells entering the WTP. However, ATSDR expanded its analysis to investigate the VOC degradation by-products, including TCE, 1,2-tDCE, and vinyl chloride.

Mr. Maslia stated the following in his expert report: "*To build further confidence in the four-level calibration for TT and to assess model uncertainty, a multiphase, multispecies finite-element model, TechFlowMP (Jang and Aral 2005, 2008), developed by ATSDR's University Partner, MESL, was run using the calibrated parameter values from MODFLOW-96 and MT3DMS (Table 7.8)*."[204] Dr. Aral provided a similar statement in his expert report: *"It also served the purpose of independent reconfirmation of the predictions of the calibrated multiphase subsurface models used by ATSDR at the Camp Lejeune site*."[205]

However, no such purpose was stated in ATSDR's reports. Instead, the multiphase-multispecies model was built to simulate the degradation of VOCs and not build confidence in the results.[206] In fact, Mr. Maslia states the following in his expert report: "*Unlike MT3DMS that simulated contaminant fate and transport in the saturated zone for a single contaminant that does not undergo degradation, TechFlowMP can simulate flow in the unsaturated zone (above the water table), in the saturated zone (below the water table), the degradation of PCE (into TCE, 1,2-tDCE, and VC), and the loss of PCE by accounting for volatilization.*"[207]

[204] Maslia (2024), Expert Report, p. 60

[205] Aral (2024), Expert Report, p.16

[206] ATSDR-TT, Ch. A, p. A41

[207] Maslia (2024), Expert Report, p. 60

The purpose of this multiphase-multispecies modeling effort, undertaken by ATSDR, is clearly stated in Chapter G of ATSDR's report on the analyses for Tarawa Terrace[208]: "*The purpose of this study is to investigate the fate, degradation, and transport of PCE and associated VOC degradation by-products - TCE, 1,2-tDCE, and VC - within the Tarawa Terrace aquifer and Castle Hayne aquifer system at and in the vicinity of Tarawa Terrace.*"

In Chapter G of ATSDR's report about the use of TechFlowMP they further explained: "*The study applies the numerical model TechFlowMP (Jang and Aral 2005b) to the Tarawa Terrace area. Calibration of the fate and transport model is based on the spatial and temporal distributions of contaminants PCE, TCE, and 1,2-tDCE at selected water supply well locations within the Castle Hayne aquifer system. Thus, the application of the TechFlowMP model was used to account for and to simulate (1) parent-daughter chain reactions, (2) multiphase environments (water and vapor), and (3) fate and transport in the unsaturated and saturated zones.*"[209]

The statements above paint a very clear picture of the intended purpose of this modeling effort. TechFlowMP was required for simulating concentrations of PCE's degradation by-products (TCE, 1,2-tDCE, and VC) and not for confirming the MT3DMS results.

Although TechFlowMP and MT3DMS simulated PCE concentrations in groundwater using the same parameters and source mass loading, there is a discrepancy in simulated PCE concentrations between TechFlowMP and MT3DMS. ATSDR indicated that "*PCE concentrations at well TT-26 using TechFlowMP are less than those using MT3DMS (Figure G6a). This is partially due to TechFlowMP simulating (1) the release of PCE from the subsurface (groundwater) to atmosphere, (2) PCE partitioning from the water phase to soil vapor phase, and (3) the placement of the contaminant source at the ABC One-Hour Cleaners site in the unsaturated and saturated zones.*"[210]

However, Robert Faye, a consultant subcontracted to ATSDR to work on the water modeling efforts, commented on this discrepancy, as well as other issues with the TechFlowMP model, in a personal communication with Mr. Maslia.[211] Regarding the PCE concentration discrepancy, Mr. Faye stated that "*[f]rom a technical point of view, I believe most or all of this unfortunate 'mess' has evolved from flawed concepts and applications on the part of GA Tech. Specifically, they applied the calibrated mass loading rate from the MT3DMS model to the unsaturated and saturated zones represented in the model.*"[212] He further indicated that "*applying the calibrated mass loading rate from the MT3DMS model to the unsaturated zone directly equates the actual ('real world') PCE loss rate at ABC One-Hour Cleaners to the MT3DMs mass loading rate. Such an equation is absurd as it does not account for retention and degradation within the unsaturated zone.*"[213] In other words, using the same mass loading rate in TechFlowMP as in MT3DMS, in both the saturated and unsaturated zones (instead of only the saturated zone as in MT3DMS), would inevitably result in lower PCE concentrations in the saturated zone.

In his personal communication with Mr. Maslia, Mr. Faye also commented on the choice of biodegradation rate. He indicated that he had "*rerun the fate & transport model with a biodegradation rate*

[208] ATSDR-TT, Chapter G, p. G4

[209] ATSDR-TT, Chapter G, p. G4

[210] ATSDR-TT, Chapter G, p. G14

[211] CLJA_WATERMODELING_01-0000075306: Email from R.E. Faye to M. Maslia, Saturday, January 13, 2007, 5:14:51 PM.

[212] CLJA_WATERMODELING_01-0000075306: Email from R.E. Faye to M. Maslia, Saturday, January 13, 2007, 5:14:51 PM. Numbered item 4.

[213] CLJA_WATERMODELING_01-0000075306: Email from R.E. Faye to M. Maslia, Saturday, January 13, 2007, 5:14:51 PM. Numbered item 4.

of 0.0005 as you required,"[214] referring to Mr. Maslia. He continued saying that "*the results are only marginally acceptable and certainly do not represent our 'best' calibration*."[215] He further stated that he "*will find it very difficult to defend these results to [his] technical peers or in a court of law*."[216]

Mr. Faye proceeded with this comment on the modeling procedure: "*I believe we have violated a fundamental rule of good modeling procedure. We let the 'tail wag the dog' and assigned extraordinary credibility to simulated numbers rather than to well established concepts. When a choice must be made between accepting less than desirable model results or violating or compromising valid conceptual models, I believe we should accept the undesirable results and explain the limitations in that context*."[217] It is not clear what Mr. Faye meant by "*desirable results*" or his admission that the modeling team assigned "*extraordinary credibility to simulated numbers than to well established concepts*."

ATSDR's report on the calculation of the biodegradation rate indicated that the value of 0.0005 was calculated based on only two concentration values at well TT-26, from samples collected on September 25, 1985, and July 11, 1991.[218] However, additional discussion was provided in the ATSDR report: *"[h]alf-lives of PCE reported in the literature range from about 360 to 720 days (Lucius and others 1990). Applying these half-lives to Equation 3 yields first-order degradation rates ranging between 0.001 and 0.002 per day, about twice to four times the rate computed using concentrations at water-supply well TT-26*."[219] Hence, Mr. Faye, the author of the report, indicated that the selected biodegradation rate in the calibrated model was low, and information in the literature would support values two to four times higher. Such a choice would result in lower PCE concentrations in the aquifer.

The above statements from Mr. Faye to Mr. Maslia indicate that the members of the ATSDR modeling team were not in agreement on important modeling aspects, resulting in discrepancies in the results between the two models (MT3DMs and TechFlowMP), and estimated concentrations that could (or should) have been lower than those calculated by ATSDR.

Regardless of the disagreement between the ATSDR modeling teams, TechFlowMP calculated PCE concentrations in groundwater that were lower than those calculated using MT3DMS. However, ATSDR chose to report the concentrations calculated by MT3DMS rather than those generated by TechFlowMP, i.e., the model they acknowledged was inclusive of all processes in the subsurface.[220] Neither ATSDR, Mr. Maslia, nor Dr. Aral, provided sufficient scientific justification for selecting the higher (MT3DMS) estimated monthly contaminant concentrations for their dose reconstruction.

Finally, similar to the selection of the K_d values for PCE (Section 4.1.2.2.2), ATSDR did not utilize site-specific data for assigning K_d values for PCE's degradation by-products. Therefore, the values used by ATSDR in TechFlowMP were lower than those estimated from site-specific data. Specifically, Table 1 provides a comparison of the K_d values used by ATSDR versus those calculated using site-specific data. Calculations were based on the discussion provided in Section 4.1.2.4.

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[214] CLJA_WATERMODELING_01-0000075306: Email from R.E. Faye to M. Maslia, Saturday, January 13, 2007, 5:14:51 PM.

[215] CLJA_WATERMODELING_01-0000075306: Email from R.E. Faye to M. Maslia, Saturday, January 13, 2007, 5:14:51 PM.

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[217] CLJA_WATERMODELING_01-0000075306: Email from R.E. Faye to M. Maslia, Saturday, January 13, 2007, 5:14:51 PM.

[218] ATSDR-TT, Chapter F, p. F28

[219] ATSDR-TT, Chapter F, p. F29

[220] ATSDR-TT, Chapter G, p. G14

Table 1. ATSDR versus Site-Specific K_d Values for Other Contaminants

Contaminant	ATSDR Value[a]	Median Value Calculated from Site-Specific Data[b]	
		All Data	Data Below 10 ft
TCE	0.1	0.17	0.13
1,2-tDCE	0.04	0.07 – 0.09	0.05 – 0.07
VC	0.003	0.1	0.08

[a] Values from Table G2, ATSDR-TT, Chapter G, p. G11

[b] Values calculated using the same assumptions as PCE. Values for log K_{oc} from ATSDR's Table D12:[221]

- TCE: 2.00
- 1,2-tDCE: 1.56 – 1.69
- VC: 1.75

Had site-specific data been used, K_d values would have been higher, as shown in Table 1. In that case, arrival times of contaminants at the supply wells would be later, and corresponding concentrations of contaminants at the wells in earlier times would be lower.

In Summary (Opinions 10 and 11): ATSDR applied two different numerical codes for modeling dose reconstruction. The results of the two codes are not in agreement. This is due, in part, to inconsistent application of contaminant source terms in the two models. Neither ATSDR, Mr. Maslia, nor Dr. Aral, provided sufficient scientific justification for selecting the higher estimated monthly contaminant concentrations for their dose reconstruction. In addition, ATSDR did not use site-specific parameter values for estimating monthly concentrations for the VOC degradation by-products. Had they used site-specific parameter values, the contaminant plumes would migrate slower to the water-supply wells, and the resulting estimated concentrations would be lower.

4.1.5 The Post-Audit Analysis Shows Consistent Bias in Model Results

A post-audit of the ATSDR model was conducted by Mr. Norm Jones of Norm Jones Consulting LLC and Dr. Jeffrey Davis of Integral Consulting Inc. (henceforth referred to as "Jones & Davis"). The post-audit included (a) extending the ATSDR model to simulate conditions through December 2008, and (b) considering pumping and monitoring data to evaluate the ATSDR's model performance "*as an interpretive and predictive tool.*"[222]

The ATSDR model simulated groundwater flow and contaminant transport during the period 1951-1994. In the post-audit, the model was extended to encompass the period 1995-2008. Selection of this timeframe was based on available pumping and monitoring data for this period. Although the post-audit report does not provide a reference for the source of these data, from my review of available documents it appears that a source was the 2018 Focused Remedial Investigation Report[223] (RI) for ABC One-Hour Cleaners.

Execution of the extended groundwater flow and contaminant transport model generated results that were post-processed by Jones & Davis to construct maps and plots for basing their opinions regarding

[221] ATSDR-TT, Chapter D, p. D15

[222] Jones & Davis, (2024), Expert Report, p. 1-2

[223] Black & Veatch (2018)

model performance. Contrary to their opinion, the extended ATSDR model does not reasonably forecast future conditions (1995 to 2008). Model simulation results were consistently biased high, overestimating observed concentrations by 2 to 3 orders of magnitude.

4.1.5.1 Post-Audit Model Results are Biased High

The Post-Audit report presents the results of the extended model and provides a discussion on model performance that is summarized in the following statements: "*In summary, the extended model demonstrates that the original model was developed using sound methods, and the model remains a reliable tool for understanding the general trends of contaminant migration in the Tarawa Terrace region. Based on this post-audit, we can find no significant evidence that would invalidate the analyses performed by ATSDR with the original model*." This statement is not supported by the data as discussed below.

Figure 17 shows Figure 6 of the Post-Audit report, illustrating comparisons between observed and simulated concentrations from the original and extended model. In the Post-Audit report it is stated that "*The points on the plot are mostly centered on the line, but as was the case with the original model, the simulated values appear to be biased on the high side, with the simulated values greater than the observed values. However, when the sites with zero observed or simulated concentrations (not shown on Figure 6) are factored in, the errors are balanced, as indicated by the low ME [Mean Error][224] (21 µg/L) reported above*."[225]

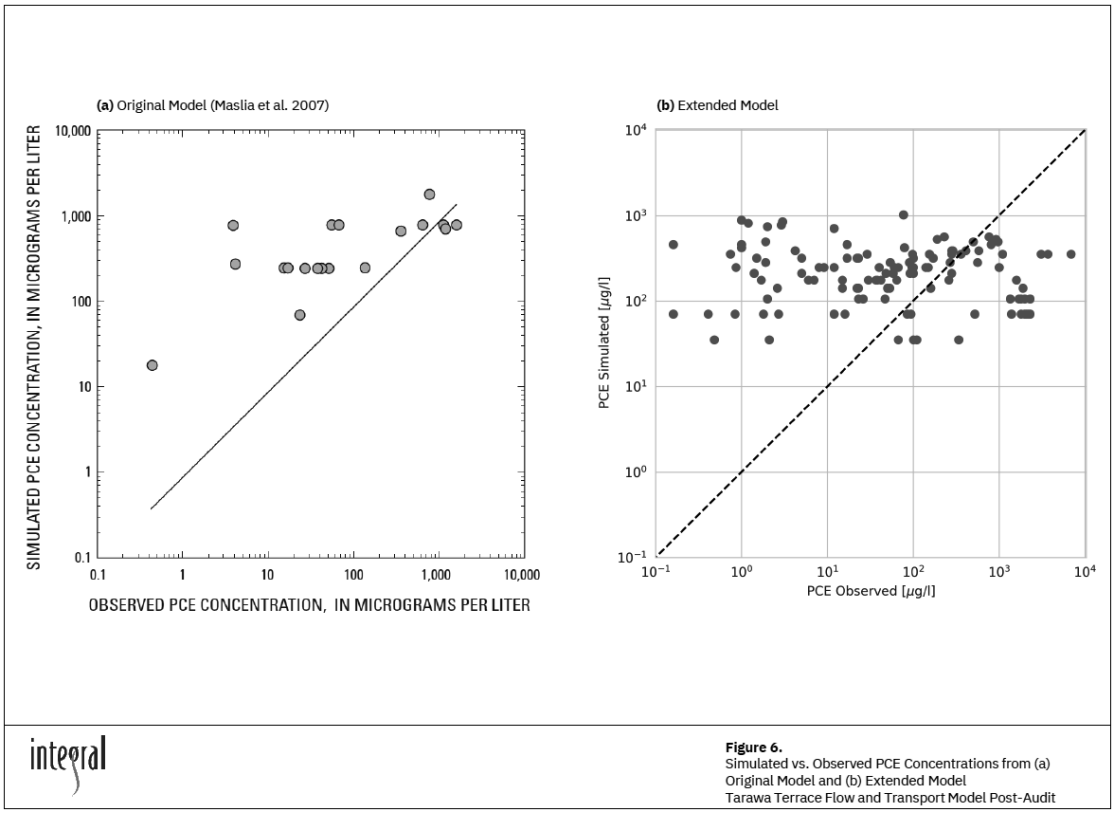

Figure 17: Observed versus Simulated PCE Concentrations from the Original and Extended Model (Integral, 2024)

[224] Note: Mean Error is the average of the differences between the simulated and observed values

[225] Jones & Davis, (2024), Expert Report, p. 5-2

Jones & Davis' statement that simulated values were higher than the observed values confirms that the model was biased high. This is consistent with ATSDR's statement about the calibrated model that "*simulated PCE concentrations moderately to substantially overpredicted observed concentrations at water-supply wells*."[226] However, their statement about the mean error is not valid, as will be illustrated below.

Observed concentrations of zero correspond to non-detections. Mean error is the difference between simulated and observed values. It can be negative or positive. A negative mean error indicates that the simulated values are lower than the observed values. A positive mean error indicates that the simulated values are higher than the observed values.

A reconstructed version of Jones & Davis' Figure 6 is shown in Figure 18, to include all zero observed and simulated concentrations. Because a value of 0.0 does not appear on a logarithmic scale, I plotted those concentrations with a surrogate value of 0.1 micrograms/L, so that all values are visible. In Figure 18, I have also included two dashed lines to indicate the bounds of the "+/- one-half order of magnitude range" that ATSDR used for evaluating its model calibration.

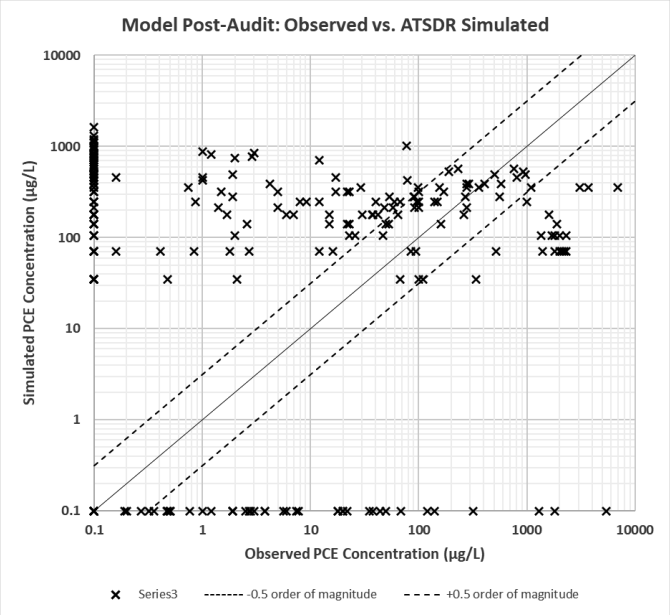

Figure 18: Scatterplot of Observed versus Simulated PCE Concentrations from the Extended Model

The following conclusions can be drawn from reviewing the results depicted in Figure 18:

- As stated in Jones & Davis' report, the simulated values are biased on the high side, being consistently greater than the observed values.
- Only a small fraction of the simulated values is within the calibration range.
- While observed concentrations vary by about 3 orders of magnitude, simulated concentrations vary by about one.

[226] ATSDR-TT, Chapter F, p. F33

- For observed non-detections (indicated by the point on the axes origin), the corresponding model simulated values (lined up along the vertical axis) vary between 106 and 1,624 µg/L. Only two simulated concentrations are lower than one hundred (35 and 71 µg/L), or two to three orders of magnitude greater than observed values.
- For all observed concentrations along the horizontal axis, spanning 4 orders of magnitude, the corresponding simulated value is zero.

The first three items above illustrate how the extended model is biased high, consistently overestimating observed data. The last two items illustrate that non-detections and simulated concentrations of zero are largely not correlated, and simulated values are biased high. For this portion of the dataset, when observed values are zero, the simulated values are greater than zero; and when simulated values are zero, observed concentrations are greater than zero. Below is a more detailed discussion on the impact of this discrepancy.

Calculation of the mean error for this portion of the dataset results in large positive errors (when the simulated value is high and the observed value is zero) and large negative errors (when the simulated value is zero and the observed value is high), balancing the total mean error. There are 125 data points for which either the observed or the simulated value are zero. The high observed values include one measurement of 5,400 µg/L, and all other values are less than 2,000 µg/L. If this extreme value is excluded from the comparison, it turns out that:

- There are 40 data points for which the observed value is greater than zero when the simulated value is zero. The mean error for this portion of the dataset is -102 µg/L. If the extreme value of 5,400 µg/L is included in the calculation, the mean error becomes -231 µg/L.
- There are 84 data points for which the simulated value is greater than zero when the observed value is zero. The mean error for this portion of the dataset is 441 µg/L.

This means that the mean error is four times higher when simulated values are greater than zero and observed values are non-detections than the reverse (i.e., when observed values are greater than zero and simulated values are zero). This further illustrates how the model overestimates the observed data and is biased high, especially when observed data indicate no presence of contamination.

To further illustrate these issues, a different plot was constructed to compare the observed and simulated concentrations from the extended model. Figure 19 shows the observed versus simulated concentrations, as well as the calibration range. In this plot, the observed data are ranked in descending order from 1998-2000 (i.e., from the highest observed value to the lowest; not listed chronologically) with their corresponding simulated values. Observed or simulated concentrations of zero are shown in this plot with a value of 0.1, so they are visible in the logarithmic scale. The first (highest) 158 ranked observed concentrations are greater than zero and the remaining 160 are non-detections. The highest observed concentration is 6,900 µg/L. Key observations from this plot are the following:

- Only a small portion of ATSDR model-calculated concentrations lie within the calibration range. All other calculated values are more than one order of magnitude greater than the observed values.
- For observed concentrations below 10 µg/L, corresponding simulated concentrations are consistently about 2 to 3 orders of magnitude greater.
- For observed non-detections, corresponding simulated concentrations are consistently 3 to 4 orders of magnitude greater.
- Only a small fraction of the simulated concentrations is lower than the corresponding observed concentrations.

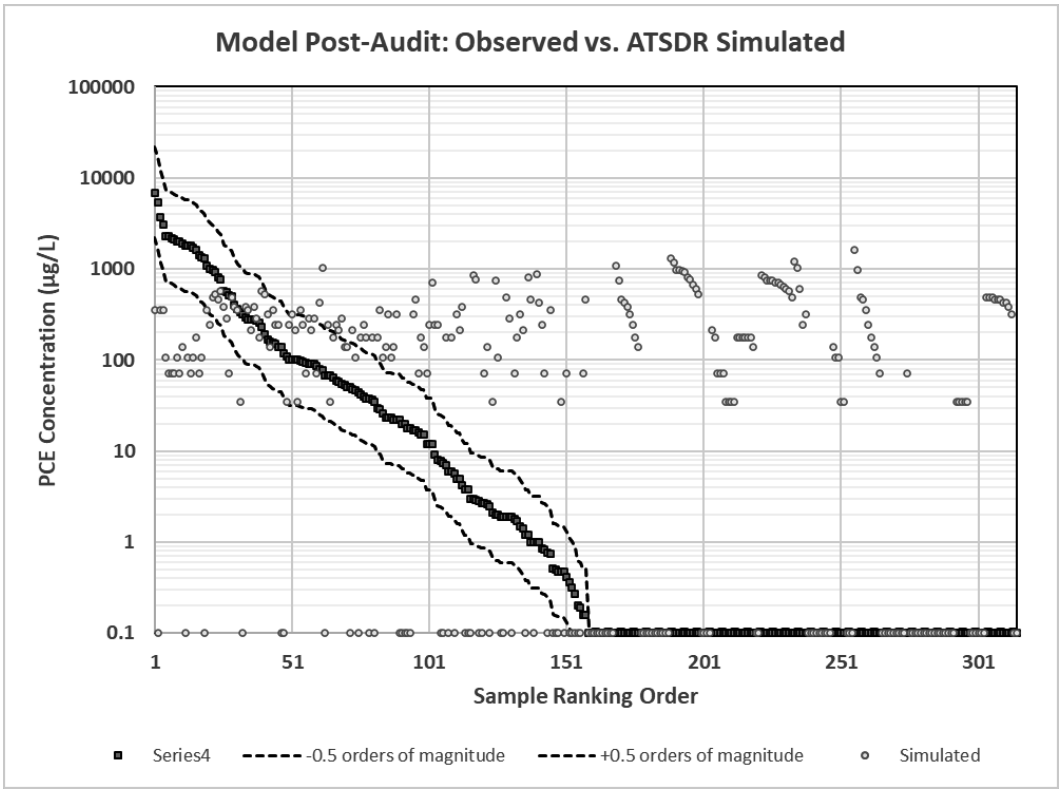

Figure 19: Observed versus Simulated PCE Concentrations from the Extended Model

Jones & Davis' report attempts to attribute these discrepancies to complex surface conditions, temporal variability, limitations in model resolution, and measurement variability.[227] While these factors are generally important, these figures clearly illustrate that the discrepancies between observed and simulated concentrations consistently and extensively exceed the generous calibration range that ATSDR defined as appropriate for evaluating the performance of the original model. If ATSDR concluded that this "+/- one-half order of magnitude range" were appropriate for the calibration of the original model, it should also be applicable to this validation dataset; and this assumption is clearly violated in the extended model.

Jones & Davis' report proceeds with a qualitative evaluation of the model results, where plume maps are constructed for different times, also showing monitoring well locations and associated discrepancies between observed and simulated concentrations at those locations. These maps are intended to illustrate that "*the spatial distribution of the errors indicates that there is a good overall agreement between the shape of the plume and the observed PCE concentrations at the monitoring wells.*"[228] However, this conclusion comes after a lengthy discussion that attempts to explain extreme discrepancies at various locations by attributing them to the factors indicated above. In this discussion, Jones & Davis state that "*the 7 wells identified as having anomalies in the observed data have high errors while the remaining 30 wells exhibit low or moderate errors, indicating good overall agreement between the simulated PCE plume and the observed concentrations over the range of the extended simulation*"[229]. The seven wells identified by Jones & Davis as having anomalous observed data are FWS-13, RWS-4A, FWC-11, C5, C13, C14, and RWC-2. Figure 20 depicts a scatterplot of observed and simulated concentrations for the other 30 wells.

[227] Jones & Davis, (2024), Expert Report, p. 5-1

[228] Jones & Davis, (2024), Expert Report, p. 5-3

[229] Jones & Davis, (2024), Expert Report, p. 5-4

The discrepancies between observed and simulated concentrations at these wells follow the same pattern as the entire set of wells. Only a small fraction of the simulated concentrations is within the calibration range, and simulated concentrations are consistently greater than corresponding observed concentrations by 2 to 3 orders of magnitude.

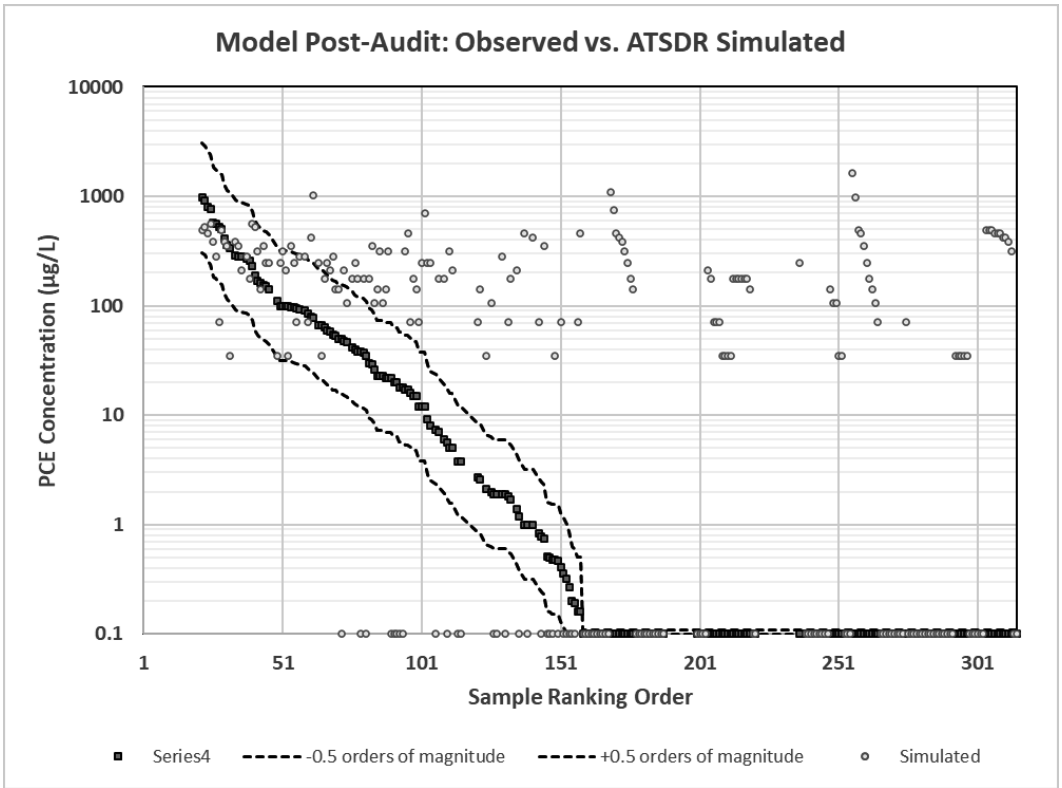

Figure 20: Observed versus Simulated PCE Concentrations from the Extended Model Excluding Wells with Alleged Anomalies in Observed Concentrations

For the reasons discussed above, the extension of the original ATSDR model does not reasonably forecast future conditions. Using data from the period 1995-2008 as a validation dataset, model simulation results were consistently biased high, overestimating observed concentrations by 2 to 3 orders of magnitude.

Jones & Davis also stated that "*it is important to qualitatively assess the overall behavior of the simulated plume in addition to quantitatively analyzing the differences in simulated and observed concentrations at specific times and locations.*"[230] However, qualitative assessment of the PCE concentrations simulated by their extended model is generally unhelpful because:

- discrepancies between observed and simulated concentrations are significant and biased;
- comparisons are drawn within a very small area, compared to the overall plume extents; and
- no data are available to evaluate whether the overall extents of the simulated plume are real.

The lack of data available to evaluate the overall extent of the simulated plume is particularly important. Plume extents and concentrations at other locations would provide supporting information for assessing model parameter values and assumptions built into the calibrated model. This directly addresses

[230] Jones & Davis, (2024), Expert Report, p. 5-1

the selection of the calibrated model parameters, and the resulting migration patterns of contamination at the site.

In Summary (Opinions 12 and 13): The extended ATSDR model (Post-Audit) shows that the model does not reasonably forecast future actual measured conditions (1995 to 2008). Model simulation results were consistently biased high, overestimating observed concentrations by 2 to 3 orders of magnitude. ATSDR's calibrated model used parameters and assumptions that results in biased-high estimate of contaminant concentrations for both the historical reconstruction and the "predictive" period of the post-audit. Therefore, the model should be recalibrated using all available data from the historical and extended period.

4.1.5.2 There are Issues with Extended Model Inputs and Post-Processed Results

Review of the Post-Audit model files also led to the discovery of certain issues related to the construction of model inputs and the presentation of model results. First, the Post-Audit report states that mass loading at the ABC One-Hour Cleaners source was set up from January 1953 through December 1983.[231] I inspected the model files and confirm that this is the timeframe implemented in the mass-loading input file constructed by Jones & Davis for the extended model. However, this timeframe is incorrect as the source in the ATSDR model was active through December 1984.[232]

In addition, Table 2 of the Post-Audit report lists the pumping wells and their operation (dates and flow rates) for the period 1995-2008. According to this table, well RWC-2 had a pumping rate of 40 gallons per minute (gpm) from 3/7/2004 to 12/16/2004. However, the input file for the well operation in the extended model shows this well with a pumping rate of 20 gpm. This is a discrepancy between Jones & Davis' modeling documentation and actual model files.

Post-processing of the model results presented in Jones & Davis' report requires clarification. In some cases, simulated concentrations corresponding to consecutive sampling events, spanning months or even years, are identical. For example, well RWC-2 is listed as having the same concentration of 106 µg/L 5 separate times between 2/1/2000 and 11/1/2002; and 6 separate times with the same concentration of 71 µg/L between 3/1/2003 and 3/1/2008. This is highly unlikely, if not impossible, in a transient model simulation.

I post-processed the model output to extract the monthly simulated concentrations at the model cell where well RWC-2 is located (confirmed by reviewing Jones & Davis' tables and the model input files). The post-processed results indicated that the simulated concentrations varied in a manner that should be expected due to the transient conditions in the aquifer and the resulting plume migration (shown by the black line in Figure 21). However, the reported concentrations by Jones & Davis (shown by the blue line in Figure 21) do not reflect that variability but, instead, appear to be constant between 2000-2002 and 2003-2008, as indicated above.

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[231] "The PCE source, which originated from ABC Cleaners and was terminated in the original model at the end of 1983, was left unchanged." Jones & Davis, 2024, p. vi; "For the transport model, PCE was introduced through a single cell corresponding to the ABC Cleaners spill location at a mass loading rate of 1,200 g/day for a period from January 1953 to December 1983" Jones & Davis (2024), p. 1-1

[232] "Mass loading occurred continuously from stress period 26 (January 1953) to stress period 408 (December 1984). Prior to stress period 25 and after stress period 408, the assigned mass loading rate was 0.0 g/d." ATSDR-TT, Chapter F, p. F25

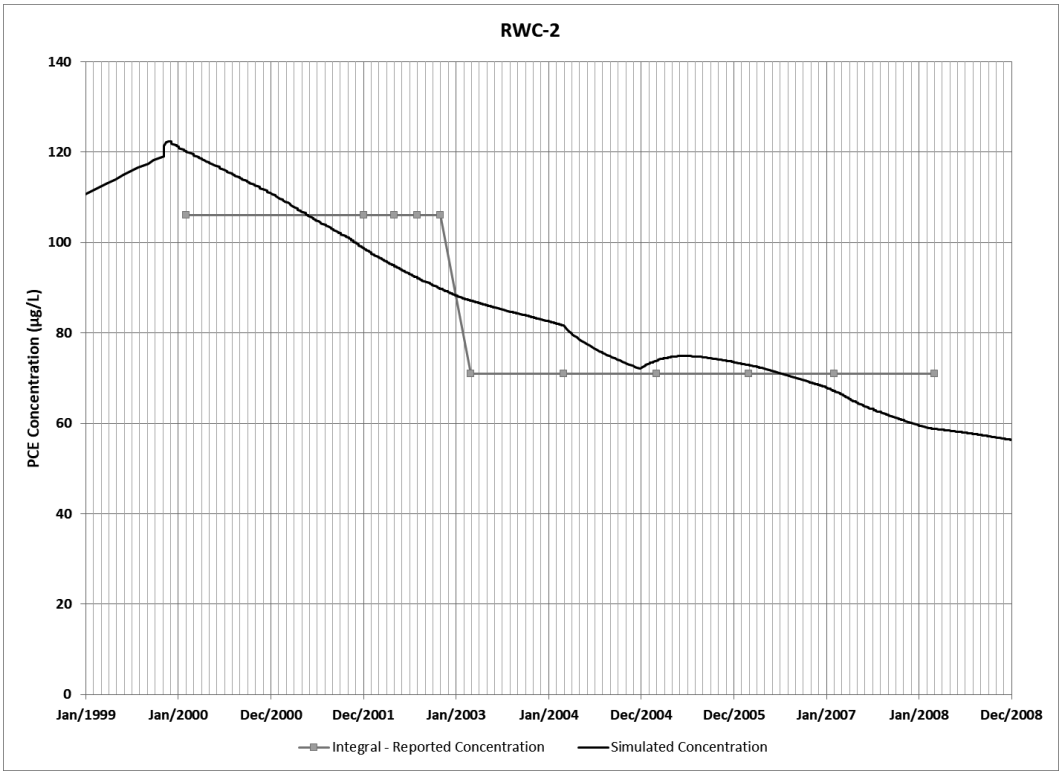

Figure 21: Observed versus Simulated PCE Concentrations from the Extended Model for Well RWC-2

In addition, tabulated and mapped reported results in the Post-Audit analysis are inconsistent in some cases. For example, in Table 5, the simulated concentration at well S8 is "<DL" on March 1, 2003. The corresponding observed value on that date is also "<DL."[233] In fact, Table 5 of the Jones & Davis report indicates that both observed and simulated concentrations at that location are always "<DL." However, Figure 9 of Jone & Davis' report shows this well in the plume interval of "greater than 5 to 50" µg/L for June 1997 (indicated by the blue arrow in Figure 22).

Similarly, Jones & Davis reported the simulated concentration at well S9 on 3/1/2003 to be "<DL." The corresponding observed value on that date is also "<DL." However, Figure 11 of Jone & Davis' report shows this well in the plume interval of "greater than 5 to 50" µg/L for March 2003 (indicated by the blue arrow in Figure 23).

[233] Note: although "<DL" means "*below detection limit*" for observed values, it is not clear what "<DL" means for simulated concentrations. See Jones & Davis, (2024), Expert Report, p. 8

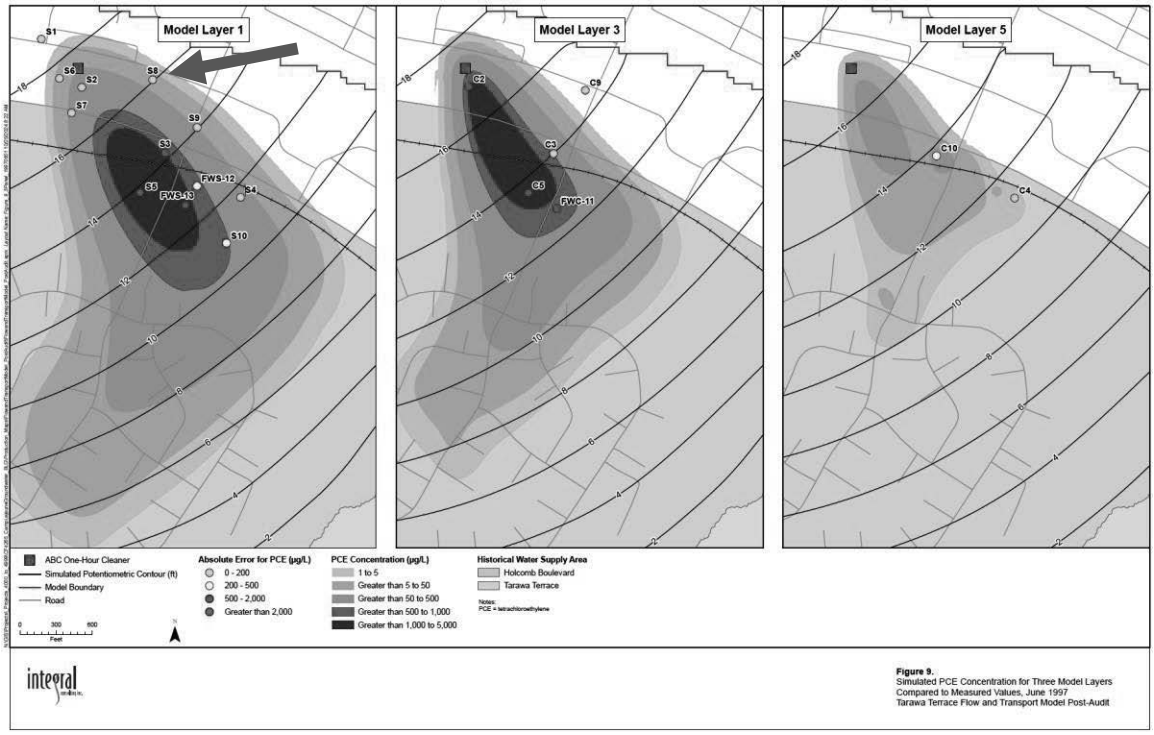

Figure 22: Simulated Plume Maps, Well Locations, and Comparison to Observed Values – June 1997 (Jones & Davis, 2024)

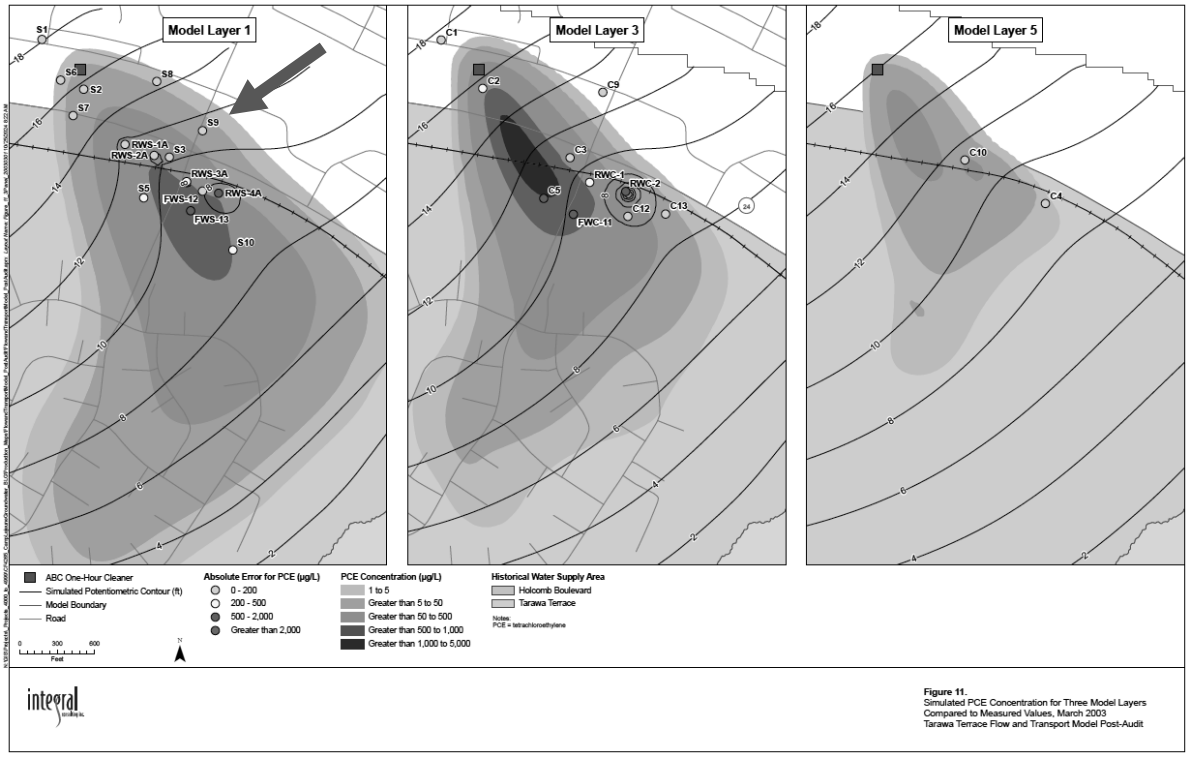

Figure 23: Simulated Plume Maps, Well Locations, and Comparison to Observed Values – March 2003 (Jones & Davis, 2024)

I post-processed the model results to confirm the simulated values for well S9, reported by Jones & Davis. Figure 24 shows my post-processed simulated concentrations at this location (black line) and the equivalent values reported by Jones & Davis (blue line). The simulated concentration at that well did not fall below 5 µg/L until after January 2006 and, therefore, Jones & Davis incorrectly reported the simulated concentration on 3/1/2003, 3/1/2004, and 3/1/2005 as "<DL," assuming that 5 µg/L was the threshold to consider a simulated value as non-detection.

Figure 24 also highlights the same issue of the questionable calculation of simulated values by Jones & Davis, which appear to be constant over different periods when the actual model results indicate otherwise. However, the reported concentrations by Jones & Davis (blue line in Figure 24) do not reflect that variability but, instead, appear to be constant between 1997-2002 and 2003-2008, as indicated above. This is highly unlikely, if not impossible, in a transient model simulation.

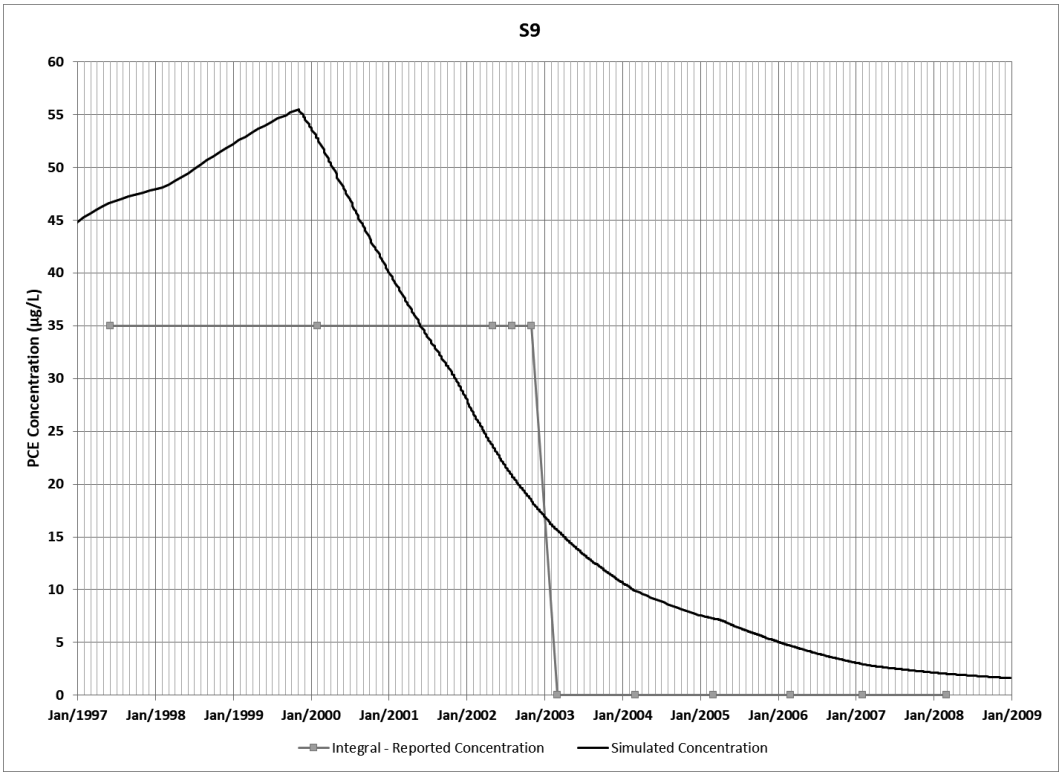

Figure 24: Observed versus Simulated PCE Concentrations from the Extended Model for Well S9

4.2 Hadnot Point – Holcomb Boulevard Area

Hadnot Point and Holcomb Boulevard are water distribution systems at Camp Lejeune served by individual water treatment plants. Hadnot Point began operations in 1942. The Holcomb Boulevard area was served by Hadnot Point until 1972, at which point it opened its own water distribution system and water treatment plant separate from Hadnot Point. For further discussion of the history of the water infrastructure at Camp Lejeune, see the expert report of Dr. Brigham.

Water quality samples taken at Camp Lejeune in the 1980s revealed contamination of VOCs. Because there were no VOC data prior to the 1980s, ATSDR attempted to use mathematical modeling to reconstruct historical concentrations of contaminants in water supply wells and at the WTPs in the absence of measured data.

ATSDR used a model to simulate historical groundwater flow and contaminant transport at Hadnot Point and Holcomb Boulevard. To construct their model, ATSDR first developed a conceptual model for groundwater flow and contaminant transport in the subsurface. To do this, they relied on past investigations at Camp Lejeune.

The groundwater flow model was created with limited available data. As with Tarawa Terrace, ATSDR also created a hypothetical well pumping schedule for Hadnot Point and Holcomb Boulevard using limited to no data. The groundwater flow model was used to create a contaminant fate and transport model, which also relied on limited data. Unlike the Tarawa Terrace model, ATSDR did not know the precise location of all contamination sources and the magnitude of contamination each source contributed. Therefore, they made arbitrary and highly uncertain assumptions to locate and quantify the contribution of these sources.

These uncertainties were highlighted in NRC's report: "*The contamination of the Hadnot Point system was more complex than Tarawa Terrace. There were multiple sources of pollutants, including an industrial area, a drum dump, a transformer storage lot, an industrial fly ash dump, an open storage pit, a former fire training area, a site of a former on-base dry cleaner, a liquids disposal area, a former burn dump, a fuel-tank sludge area, and the site of the original base dump*.."[234]

The Hadnot Point and Holcomb Boulevard areas of the base separately provided water to consumers using different water distribution networks. However, the Holcomb Boulevard and Hadnot Point water distribution networks could be connected if and when necessary to address water supply needs in Holcomb Boulevard. Usually, this occurred during the summer months when water demand at Holcomb Boulevard was high. On one occasion, Hadnot Point provided all water to the Holcomb Boulevard water distribution system for ten days between January 27, 1985, and February 7, 1985. This was because the Holcomb Boulevard system had to be flushed due to the presence of benzene (see details in Hennet, 2024).

For Hadnot Point, as with Tarawa Terrace, ATSDR assumed concentrations of contaminants in the influent to the WTP were equal to the concentrations of contaminants in the "finished water" that was delivered to consumers.[235] This assumption is incorrect, as treatment of the influent to the treatment plant

[234] NRC (2009), p. 5

[235] ATSDR stated that "*for this study, finished water is defined as groundwater that has undergone treatment at a water treatment plant and was subsequently delivered to a family housing unit or other facility. Throughout this report and the Hadnot Point–Holcomb Boulevard report series, the term finished water is used in place of terms such as finished drinking water, drinking water, treated water, or tap water*." (ATSDR-HP, Chapter A, Supplement 6, p. S6.21) However, ATSDR used simulated contaminant concentrations in the influent to the WTP to calculate concentrations in the water delivered to a family housing or other facility, without considering any contaminant losses during treatment. This was an important assumption of significant impacts, as discussed by Hennet (2024). Therefore, references to historical reconstruction of VOC concentrations hereafter are associated

resulted in evaporative and other losses, reducing contaminant concentrations in the "finished" water. Unlike Hadnot Point and Tarawa Terrace, the Holcomb Boulevard water distribution system itself was not contaminated. However, water quality in the Holcomb Boulevard water distribution network was impacted by the addition of contaminated water from Hadnot Point during interconnection events.

To account for contamination entering the Holcomb Boulevard water distribution network during interconnection events, ATSDR developed a model to simulate water flow in the piping network and the resulting migration of contamination throughout.

Based on my professional judgment, there was insufficient data to conduct groundwater flow and contaminant transport model calibration and uncertainty analysis. In fact, ATSDR admitted that *"[f]or the HPHB study area, data were extracted to compile operational chronologies for nearly 100 supply wells compared to only 16 water-supply well operational chronologies for the TT study area. The substantive increase in the numbers of data values for the HPHB study area compared to the TT study area (Table A2) are indicative of the increase in complexity and difficulty of reconstructing historical contaminant concentrations for the HPHB study area. This point is further highlighted given the multiple source contaminants (3) and numerous contaminant source areas (23) requiring identification and characterization for the HPHB study area as described in Faye et al. (2010, 2012) and discussed in subsequent sections of this report.*"[236]

Given the fact that prior to 1982, no water quality data were available, the resulting calibrated model was highly uncertain. ATSDR's sensitivity and uncertainty analysis evaluated a range of parameter values, some of which, when compared to site-specific data, did not reflect the site conditions. In addition, ATSDR used extreme values for some parameters in their sensitivity analysis. ATSDR then used the results of this sensitivity analysis to draw conclusions on the range of historical concentrations at the influent to the HP WTP.

Additionally, ATSDR's calibrated model is highly uncertain with regard to the start date and magnitude for mass loading from the different contamination sources. As ATSDR indicated in the Hadnot Point–Holcomb Boulevard reports, the model is sensitive to the start date for the release of contamination at Hadnot Point, which could have varied by several years.[237] This would significantly impact the timing and magnitude of concentrations at the water supply wells, and therefore, the water treatment plant. ATSDR focused on the period of interest to the epidemiological studies (1968-1975) but ignored the impact of this uncertainty for years prior to 1968[238].

An uncertainty analysis should provide a range of potential model outcomes that envelops the calibrated model. The calibrated model should generally sit in the middle of the uncertainty range. ATSDR's calibrated model sits at the top of or above the uncertainty range when the potential variability of historical pumping operations was evaluated. This demonstrates that the calibrated model was biased high.

NRC provided recommendations to address modeling hurdles: "*Because any groundwater modeling of the Hadnot Point system will be fraught with considerable difficulties and uncertainties, simpler modeling approaches should be used to assess exposures from the Hadnot Point water system.*

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with concentrations in the influent to the treatment plant, and not after-treatment "finished" water that entered the water distribution network.

[236] ATSDR-HP, Chapter A, p. A10

[237] "Variations in source-release dates of ±9 years show MCL exceedance-date variations of about 5 years earlier to 14 years later than the calibrated TCE MCL exceedance date (August 1953)." ATSDR-HP, Chapter A, p. A84

[238] "In terms of historical reconstruction results of interest to the ATSDR epidemiological studies (finished-water concentrations of TCE during the period 1968–1985), the variation (and uncertainty due to a lack of data) in source-release dates does not appear to have a substantial effect." ATSDR-HP, Chapter A, p. A84

Simpler modeling will not reduce the uncertainty associated with the estimates, but they have the advantage of providing a broad picture of the timeframe and magnitude of exposure encountered by people who used water from that system more quickly and with less resources than complex modeling exercises."[239] However, as will be shown in the following sections:

- ATSDR implemented the same complex modeling approach as in Tarawa Terrace for PCE and TCE, and only simplified their approach for modeling the VOC degradation by-products;
- The resulting calibrated model was riddled with uncertainties; and
- ATSDR performed limited sensitivity and uncertainty analyses that did not provide any confidence in the calibrated model results or their uncertainty range.

Similar concerns exist for the ATSDR's treatment of the degradation by-products of PCE and TCE (1,2-DCE and VC) and benzene.

4.2.1 Available Data are Limited or Non-Existent

To construct the groundwater flow model, ATSDR used available data, which included:

- Horizontal hydraulic conductivities from more than 200 aquifer and slug test analyses;[240]
- Aquifer specific yield and storativity values, based on data from the Tarawa Terrace model;[241] and
- Precipitation data from the Maysville-Hoffman Forest station with records from 1951-1994.[242]

Pumpage information at individual supply wells was not available for the study period. ATSDR developed assumed well pumping schedules and flow rates through a complex process. To do that, they relied on ancillary data.[243]

To construct the contaminant transport model, ATSDR used model parameters that were based on a literature review and the professional judgment of the modelers.

Contaminant concentration data at water supply wells were available in 1984-1985 and up to 1991. Eight samples were collected from water-supply wells after 1991, and up to 2005.[244] Similar to Tarawa Terrace, observed data from the historical period are unavailable prior to 1982. The HP WTP was built in 1942.[245] This means there was a forty-year period for which there is no historical water quality data that could be used to inform the model calibration. Figure 25 depicts ATSDR's Figure 18[246] with an added, yellow-highlighted area to illustrate the historical period of no available data. Appendix E lists the sampling data from the water supply wells and WTP available to ATSDR.

[239] NRC (2009), p. 5

[240] ATSDR-HP, Chapter A, Supplement 4, p. S4.5

[241] ATSDR-HP, Chapter A, Supplement 4, p. S4.18

[242] ATSDR-HP, Chapter A, Supplement 4, p. S4.18

[243] ATSDR-HP, Chapter A, Supplement 2, p. S2.2

[244] ATSDR-HP, Chapter A, pp. A21-22

[245] ATSDR-HP, Chapter A, p. A10

[246] ATSDR-HP, Chapter A, p. A46

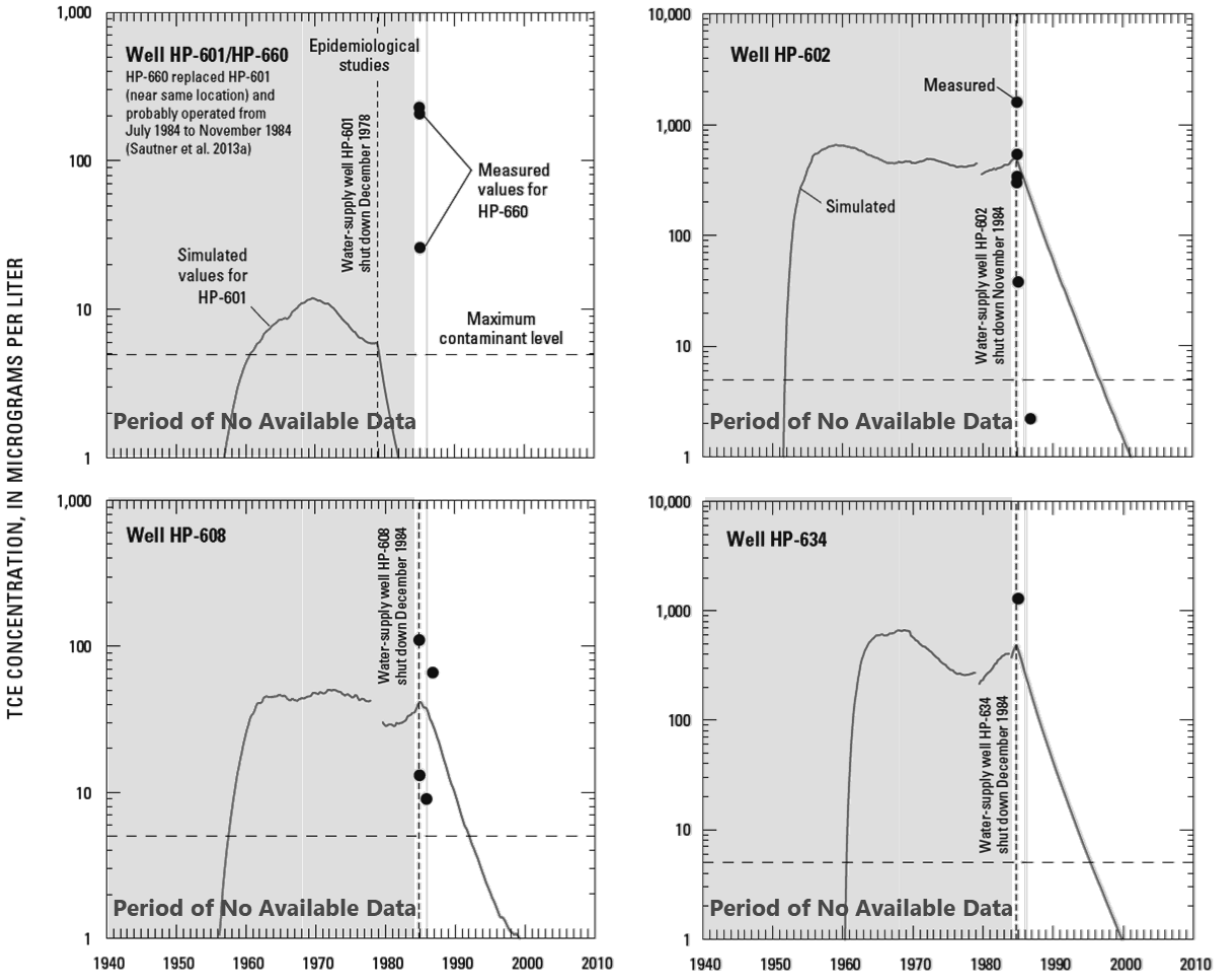

Figure A18. Reconstructed (simulated) and measured concentrations of trichloroethylene (TCE) at selected water-supply wells within the Hadnot Point Industrial Area, Hadnot Point–Holcomb Boulevard study area, U.S. Marine Corps Base Camp Lejeune, North Carolina. Groundwater-flow simulation using MODFLOW (Harbaugh 2005) and contaminant fate and transport simulation using MT3DMS (Zheng and Wang 1999). (See Figure A13 for well locations.)

Figure 25: Period of No Available Data for ATSDR's Model Reconstruction in Hadnot Point

In Summary (Opinion 14): ATSDR relied on very limited data for constructing the groundwater, fate, and transport models for dose reconstruction in Hadnot Point. Selection of model parameters was based, primarily, on professional judgment.

4.2.2 Pumping Reconstruction is Highly Uncertain

As part of its historical reconstruction analysis, ATSDR had to generate an input file for the groundwater flow model to incorporate the historical operation of water supply wells. This is important because well operation impacts groundwater flow and contaminant transport. The model simulates monthly average conditions. Therefore, to develop this input, operational patterns (including hours of operation and associated pumping rates) would be necessary to calculate monthly-average pumping rates for all wells in the HPHB area for the historical period. However, no such data were available.

ATSDR made arbitrary assumptions to reconstruct the pumping history for each well. ATSDR provided the following discussion regarding data availability:

- "*Detailed daily data pertaining to the pumping schedule of the wells are available subsequent to January 1998 (Scott R. Williams, U.S. Marine Corps Base Camp Lejeune, written communication, December 2008).*
- *Prior to 1998, data pertaining to well operations are limited or unavailable.*
- *Similarly, daily WTP raw-water volumes are available after December 1994.*
- *Between 1980 and 1994, monthly raw-water volumes are available;*
- *[Y]early volumes are available for some years prior to 1980. A trendline was used to estimate raw-water flows for years prior to 1980 when no data exist.*"[247]

This information informed ATSDR's assumptions on monthly flows for the period 1980 to 2004. Based on that, ATSDR implemented a complex procedure for calculating monthly flows for the period prior to 1980. ATSDR provided the following discussion regarding this process:

- "*Monthly raw-water flow percentages were then calculated using known monthly data for the period 1980–2004.*
- *These values were used to estimate monthly raw-water flows prior to 1980. This methodology is based on two assumptions:*
- *(1) similar characteristics of the operational patterns of the wells and WTPs for the periods of time before and after January 1998, and*
- *(2) equality between total water volume delivered to the WTP from the operating wells and the WTP raw-water volume data at all times*."[248]

Hence, for the historical period, ATSDR developed uncertain estimates of water flows to the treatment plant based on sparse data and assumptions riddled with uncertainty.

Once ATSDR developed arbitrary and uncertain estimates of WTP raw-water flows for the period 1942 to 1980, they proceeded with developing a complex process for reconstructing a similarly arbitrary and uncertain operation of pumping wells in the HPHB area for the historical period 1942 to 1997. As part of this process, ATSDR had to rely on the following ancillary data: "*(1) daily operational records, January 1998–June 2008 (Scott R. Williams, U.S. Marine Corps Base Camp Lejeune, written communication, December 2008), (2) Camp Lejeune Historic Drinking Water Consolidated Document Repository records (CLHDW CDR 2011), (3) Camp Lejeune Water Documents (CLW 2007), and (4) U.S. Geological Survey (USGS) well inventory documents (USGS, well inventory, written communication, March 2004).*"[249] The ancillary data they used did not include information specific to the well operation and associated pumping rates.

As part of the reconstruction process, ATSDR used data and assumptions from the period 1998 to 2008 (the "training period") to construct operational patterns for the water supply wells that were active during that period. This assumed operational pattern during the "training period" informed ATSDR's complex reconstruction process for the historical period 1942-1997. Thus, it was assumed that a well would

[247] ATSDR-HP, Chapter A, Supplement 2, p. S2.2

[248] ATSDR-HP, Chapter A, Supplement 2, p. S2.2

[249] ATSDR-HP, Chapter A, Supplement 2, p. S2.2

be operated in the historical period based on a pattern similar to the more recent "training period," with further adjustments to account for information on the varying capacity[250] of the wells, where available.

However, to further complicate the process, several wells from the historical period (1942-1998) were not in service during the "training period." For those wells, historical reconstruction of their operational patterns was based on other "surrogate"[251] wells with characteristics that ATSDR considered similar to those wells without available data. The wells not in service during the training period were assumed to have operated in the historical period under the same operational patterns[251] as their corresponding surrogate wells. This is yet another significant assumption, as the historical operational patterns were unknown and "surrogate" well operation introduced additional uncertainty to the historical reconstruction.

Figure 26 lists the pumping wells not in service during the training period, and their corresponding surrogate wells identified by ATSDR. In this figure, select groups of wells are indicated for which operational patterns for the historical period were based on the same surrogate well, to illustrate how extensive this process was and the limited amount of information that was available to reconstruct historical well operations. For example, the operation of well HP-634 for the period 1942-1998 was based on well HP-606, as was the case for wells HP-601, HP-603, HP-608, and HP-660. The operation of another 8 wells (HP-602, HP-604, HP-605, HP-607 (old), HP-630, HP-635, HP-636, and HP-637) was based on well HP-642. Similarly, HP-633 was used as a surrogate for the operation of wells HP-610, HP-651, and HP-653.

The operation during the training and historical period for the three surrogate wells HP-606, HP-633, and HP-642 is illustrated in Figure 27, Figure 28, and Figure 29, respectively. These figures demonstrate how limited information was available for reconstructing past operations. They also demonstrate how arbitrary this reconstruction was, considering that only limited data were available for any individual well. Yet, ATSDR estimated monthly operational histories for each well based on several highly uncertain assumptions. It is also important to note that well operation during the training period addressed different requirements for raw-water delivery to the WTP compared to the historical period and, therefore, "lessons-learned" from one period cannot be directly transferred to another.

ATSDR developed this complex reconstruction process for defining monthly pumping rates of all wells in the HPHB area, based on arbitrary assumptions for schedules of operation, well conditions, and operator choices for which wells to operate at any given time. This injects a high degree of uncertainty into the HPHB model.

[250] The capacity of a well varies with time, due to clogging of the well screen (accumulation of sediments, mineral deposits, or growth of microorganisms), infrastructure issues (pump efficiency), among other (Glotfelty, 2019)

[251] ATSDR-HP, Chapter A, Supplement 2, p. S2.13

Table S2.2. Identification of surrogate water-supply wells used to represent untrained wells, Hadnot Point–Holcomb Boulevard study area, U.S. Marine Corps Base Camp Lejeune, North Carolina.

[Refer to Figure S2.1 for water-supply well locations and Figure S2.2 for operational chronologies]

Water-supply wells	
Untrained	**Surrogate**
Holcomb Boulevard water treatment plant service area	
HP-645	HP-644
HP-649	HP-648
HP-706	HP-705
HP-707	HP-704
Hadnot Point water treatment plant service area	
HP-601	HP-606
HP-602	HP-642
HP-603	HP-606
HP-604	HP-642
HP-605	HP-642
HP-607 (old)	HP-642
HP-608	HP-606
HP-609	HP-628 (new)
HP-610	HP-633
HP-611 (old)	HP-623
HP-612 (old)	HP-607 (new)
HP-614 (old)	HP-622
HP-615	HP-616
HP-617 (old)	HP-654
HP-618 (old)	HP-641
HP-619 (old)	HP-620
HP-621 (old)	HP-629 (new)
HP-624	HP-628 (new)
HP-625	HP-662
HP-626	HP-628 (new)
HP-627 (old)	HP-661
HP-628 (old)	HP-662
HP-629 (old)	HP-640
HP-630	HP-642
HP-631	HP-652
HP-634	HP-606
HP-635	HP-642
HP-636	HP-642
HP-637	HP-642
HP-638	HP-628 (new)
HP-639 (new)	HP-662
HP-639 (old)	HP-662
HP-651	HP-633
HP-653	HP-633
HP-655	HP-662
HP-660	HP-606
LCH-4006	LCH-4009
M-1	LCH-4009
M-2	LCH-4007

Figure 26: ATSDR Surrogate Water-Supply Wells

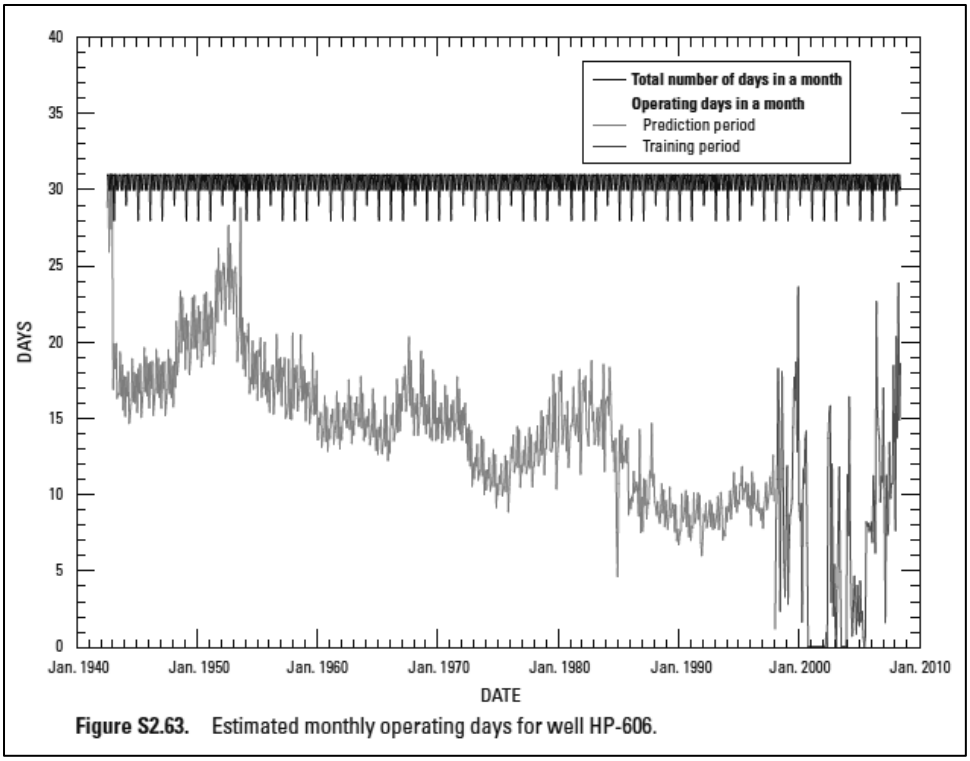

Figure 27: Training and Prediction Period Operation of Surrogate Water-Supply Well HP-606

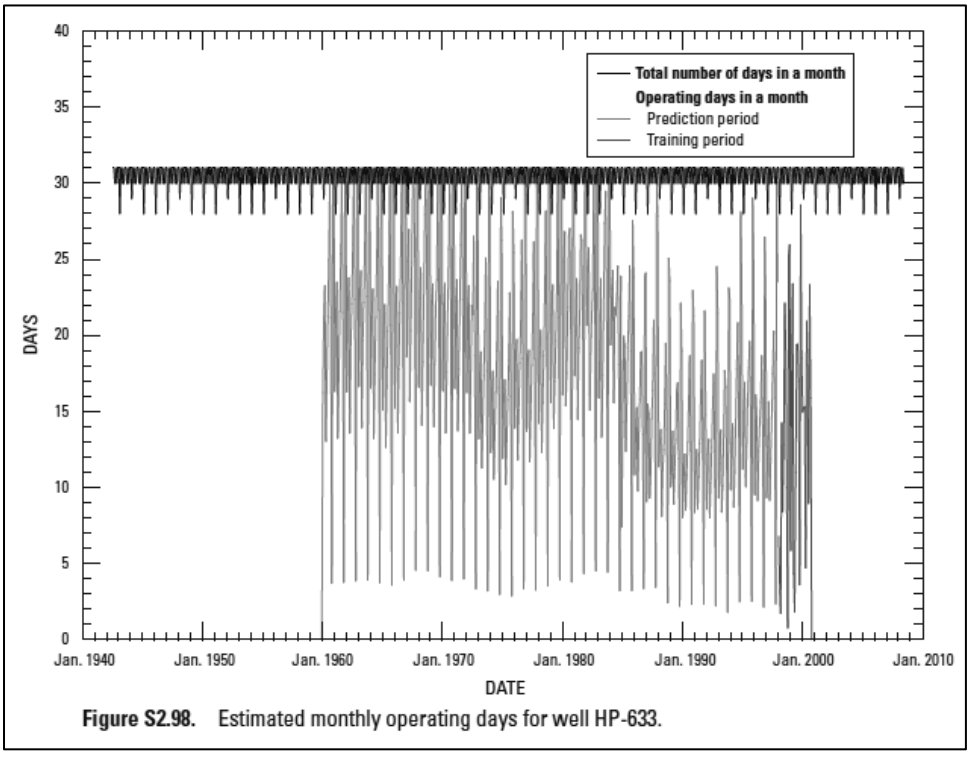

Figure 28: Training and Prediction Period Operation of Surrogate Water-Supply Well HP-633

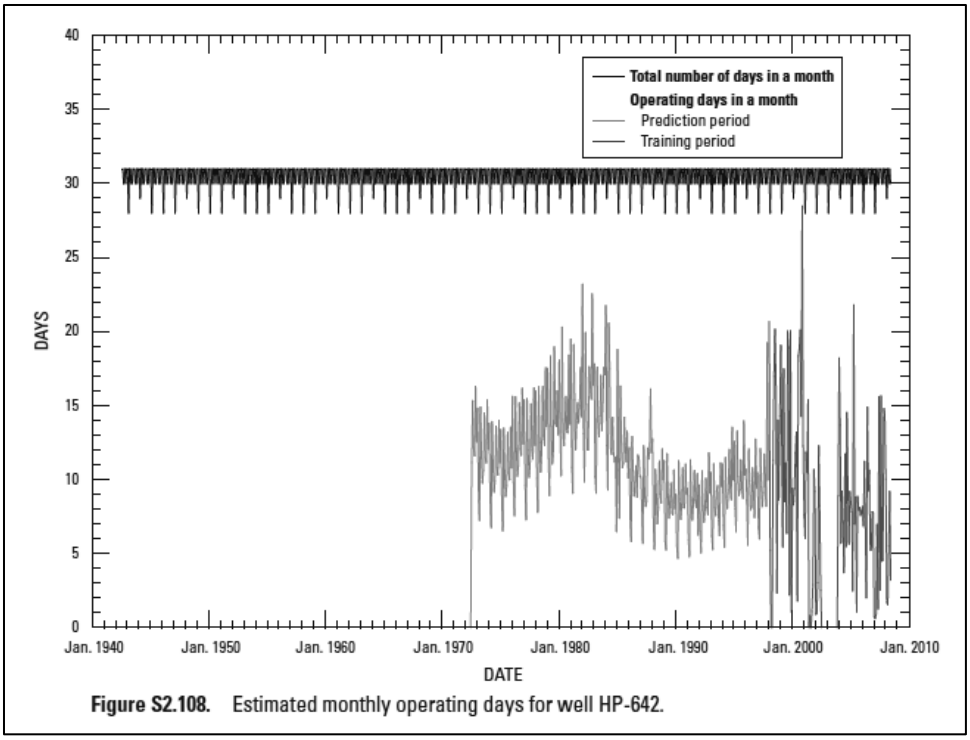

Figure S2.108. Estimated monthly operating days for well HP-642.

Figure 29: Training and Prediction Period Operation of Surrogate Water-Supply Well HP-642

In Summary (Opinion 14): ATSDR relied on very limited data for constructing its model for Hadnot Point. Selection of model parameters was based, primarily, on professional judgment. ATSDR made arbitrary assumptions and implemented a complex methodology with significant inherent uncertainty to reconstruct the pumping history for each well.

4.2.3 ATSDR's Model Calibration was Based on Limited Data and was Biased High

ATSDR conducted its calibration of the HPHB transient groundwater flow model relying on a very limited data set. As ATSDR admitted, "*only a few monitor wells in the study area contained continuous water-level data useful for assessment and calibration of the transient model*."[252] In addition, the variability in hydrogeologic features in the subsurface of the model was identified by ATSDR as a limitation.[253] as it was based on limited geophysical data. As a result, the variability in aquifer stratigraphy was consolidated in the model layer structure, simplifying aquifer complexity.

Recall that the monthly pumping rates of all wells in the model were developed using a complex reconstruction process based on arbitrary assumptions. These rates were fixed but uncertain, and underpinned model performance. Aquifer parameters were adjusted during model calibration to fit changes in water-level data caused by changes in pumping rates, among other factors.

The contaminant transport model was constructed based on numerous assumptions on parameter values as site specific data were limited or nonexistent. This problem was exacerbated by the fact that the assignment of source-release location, timing, and mass loading were unknown.[254] These quantities were

[252] ATSDR-HP, Chapter A, Supplement 4, p. S4-24

[253] ATSDR-HP, Chapter A, Supplement 4, p. S4-34

[254] ATSDR-HP, Chapter A, Supplement 6, p. S6.45

arbitrarily assigned to the model. As a result, forty years of aquifer conditions and potential contamination, after 1942, were calibrated to limited water quality data available starting in 1982. Therefore, a statistical evaluation of the contaminant transport model calibration could not be performed due to insufficient water quality data.

In one instance, model calibration was improperly influenced by the interpretation of sampling data. Erroneous concentrations reported for well HP-634 were used in the calibration, while non-detections were ignored.

A discussion on ATSDR's calibration of the groundwater flow and contaminant transport models is provided below.

4.2.3.1 The Groundwater Flow Model has Significant Limitations in the Absence of Data for Calibration

ATSDR calibrated its groundwater flow model combining a trial-and-error approach with automated parameter estimation.[255] The steady-state model, constructed for simulating pre-development conditions (i.e., ambient groundwater flow in the absence of pumping), was calibrated using more than 700 water-level measurements.[256] The transient-state model simulated conditions from January 1942 to June 2008, and was calibrated using a much smaller dataset as "*only a few monitor wells in the study area contained continuous water-level data useful for assessment and calibration of the transient model*"[257]

For the calibration of the groundwater flow model, ATSDR provided a discussion[258] addressing key issues related to the available data and model limitations. In the introductory paragraph of the discussion, ATSDR stated the following: "*Analyses and interpretations of the groundwater-flow model results should be considered in the context of model limitations and accuracy of water-level data. Results from the calibrated groundwater-flow model are used to estimate contaminant concentrations in groundwater; therefore, it is also important to consider the accuracy of the flow model results in the context of contaminant fate and transport results*."[259]

Recall that the subsurface is composed of stratified units, each with different characteristics that must be accounted for in the model. The variability in hydrogeologic features in the subsurface of the model was identified by ATSDR as a limitation, as it was based on limited geophysical data: "*the thickness of hydrogeologic units should be considered an approximation.*"[260] ATSDR continued, stating that "*multiple hydrogeologic units were combined into multiple layers. For example, layers 1 and 5 contain multiple hydrogeologic units (Table S4.1). However, contaminant transport model results typically are more sensitive than groundwater-flow model results to the combination of multiple hydrogeologic units in a model layer.*"[261] This means that ATSDR consolidated multiple geologic features into single layers of the model. This impacts the simulation of contaminant transport in the aquifer, making it inherently uncertain.

ATSDR identifies additional model limitations associated with model features such as drains, specified head boundaries, or no-flow boundaries, given that no historical data were available to properly represent these features in the model, adding to model uncertainty.[262]

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[255] ATSDR-HP, Chapter A, Supplement 4, p. S4.21

[256] ATSDR-HP, Chapter A, Supplement 4, p. S4-24

[257] ATSDR-HP, Chapter A, Supplement 4, p. S4.24

[258] ATSDR-HP, Chapter A, Supplement 4, p. S4.34

[259] ATSDR-HP, Chapter A, Supplement 6, p. S6.45

[260] ATSDR-HP, Chapter A, Supplement 6, p. S6.45

[261] ATSDR-HP, Chapter A, Supplement 6, p. S6.45

[262] ATSDR-HP, Chapter A, Supplement 4, p. S4.35

ATSDR also concluded that parameter estimation was successful in determining hydraulic conductivities in layers 1, 3, and 5.[263] However, ATSDR also stated that "*[m]ore rigorous sensitivity analyses could be conducted by computing the covariance matrix. However, initial simulations indicated that computing the covariance matrix using parameter estimation was time prohibited in terms of computational times using equipment available to authors at the time model calibration and sensitivity analyses were conducted.*"[264] This statement indicates that a more rigorous sensitivity analysis would be warranted to ensure that the estimated parameters were within reasonable bounds, but that it was never conducted.

In Summary (Opinion 14): ATSDR relied on very limited data for calibrating its flow model for Hadnot Point. ATSDR identified significant model limitations associated with model features, and especially geologic representations. The lack of calibration data and the identified model limitations had substantial impacts on the quality of ATSDR's model calibration.

4.2.3.2 The Contaminant Transport Model is Constructed Using Uncertain Assumptions

For the calibrated contaminant transport model, ATSDR stated that "*insufficient water-quality data existed to conduct a statistical analysis for assessment of model calibration fit*."[265] ATSDR therefore admitted that it could not perform a quantitative assessment of the model calibration fit. This would require water-quality data from the historical period to constrain the calibration. As a result, only a qualitative assessment of the model calibration was performed.

This problem was further exacerbated by the following admission by ATSDR: "*specific data pertinent to the timing of initial deposition of contaminants to the ground or subsurface, chronologies of waste-disposal operations, such as dates and times when contaminants were deposited in the HPLF, or descriptions of the temporal variation of contaminant concentrations in the subsurface generally are not available*."[266] ATSDR therefore acknowledged that critical information on contaminant sources was unavailable. Recall NRC's critique that "*any groundwater modeling of the Hadnot Point system will be fraught with considerable difficulties and uncertainties.*"[267]

Despite this critical lack of data, ATSDR proceeded with model construction and calibration. According to ATSDR, *"Determining these types of source identification and characterization data became part of the historical reconstruction process, whereby the contaminate fate and transport model was used to test source locations, varying concentrations, and beginning and ending dates for leakage and migration of source contaminants to the subsurface and the underlying groundwater-flow system*."[268] ATSDR therefore admitted that source-release locations, timing, and mass loading were unknown, and these quantities were arbitrarily assigned to the model in order to fit the limited water-quality data available starting at 1982.

For example, ATSDR assigned the same, constant TCE concentration at all sources in the HP Industrial Area. Specifically, as illustrated in Figure 30, ATSDR assumed that contamination released from the source at Building 1601 resulted in TCE concentrations across three layers in the aquifer at that location

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[263] "*Therefore, parameter estimation was appropriate for the steady-state model calibration phase to determine recharge and hydraulic conductivity for layers 1, 3, and 5*." ATSDR-HP, Chapter A, Supplement 4, p. S4.35

[264] ATSDR-HP, Chapter A, Supplement 4, p. S4.35

[265] ATSDR-HP, Chapter A, Supplement 6, p. S6.45

[266] ATSDR-HP, Chapter A, Supplement 6, p. S6.45

[267] NRC (2009), p.5

[268] ATSDR-HP, Chapter A, Supplement 6, p. S6.45

which were equal to 640,000 µg/L on a monthly basis for a period of 42 years.[269] This value is arbitrary and unverifiable, as no site-specific data are available to support reconstructing the history of releases or quantifying their associated magnitude. The assumption that a constant mass loading of the same magnitude occurred at all sources for more than 40 years is highly uncertain and indicates the type of assumptions that were made for the historical reconstruction.

Table S6.5. Calibrated contaminant fate and transport model parameter values used to describe contaminant sources in the Hadnot Point Industrial Area (HPIA) and Hadnot Point landfill (HPLF) area, Hadnot Point–Holcomb Boulevard study area, U.S. Marine Corps Base Camp Lejeune, North Carolina.

Source area[1]	Cell location (row, column, layer)[2]	Concentration, in milligrams per liter	Source duration
Trichloroethylene (TCE)			
Building 1601	165, 116, 1–3	640	January 1951–June 1993
Building 900 area			
Source 1	102, 178, 1–3	640	January 1957–December 1994
Source 2	108, 179, 1–3		
Source 3	113, 173, 1–3		
Building 1401	122, 138, 1–3	640	January 1951–June 1993
Building 1115	145, 121, 1–3	640	January 1951–June 1993
Landfill area			
Source 1	159, 156, 1–7	256–384	January 1948–June 2008
Source 2	154, 145, 1–7		
Tetrachloroethylene (PCE)			
Landfill area			
Source 1	159, 156, 1–5	16–105	January 1948–June 2008
Source 2	154, 145, 1–5	6–42	
Benzene			
Building 1601			
Source 1	168, 117, 1	1.7	January 1951–June 1993
Source 2	171, 113, 1		

[1] Refer to Figures S6.2 and S6.3 for maps showing location

[2] Cell location corresponds to their respective models (i.e., HPIA or HPLF). Cell location with coordinates row 1, column 1 and layer 1 corresponds to the northwest corner and uppermost cell of the total model domain

Figure 30: Contaminant Sources in the HPHB Transport Model (ATSDR, Table S6.5)

ATSDR also assigned TCE sources at Buildings 1115 and 1401 (Figure 30). Unlike other sources, for which ATSDR provided supporting information regarding their location and nature, for these two buildings ATSDR indicated the following: "*TCE releases around Buildings 1115 and 1401 have been documented to a lesser degree*."[270] ATSDR continued indicating that "*[t]he presence of chlorinated alkenes around Building 1115 is documented by Faye et al. (2012, Table D5), and the concentrations varied from below detection limits to maximum values of 160 µg/L for TCE, 11 µg/L for PCE, 110 µg/L for total DCE, and 6 µg/L for VC. The chlorinated alkenes found around Building 1115 are presumably the result of natural attenuation of TCE*."[271] All this is to say that ATSDR relied on limited information about the presence and extent of contamination at those locations and assigned source releases of which the timing and magnitude were arbitrary.

[269] ATSDR-HP, Chapter A, Supplement 6, p. S6.17

[270] ATSDR-HP, Chapter A, Supplement 6, p. S6.6

[271] ATSDR-HP, Chapter A, Supplement 6, p. S6.6

With regard to transport parameters in the model, and particularly sorption, ATSDR stated that "*[s]orption in the HPHB study area is assumed to be similar to sorption in the TT study area of USMCB Camp Lejeune described in Faye (2008)*."[272] However, as indicated in the discussion for the Tarawa Terrace model, there are important differences in the calculation of the sorption-related parameters between the two models. In the HPHB model, the value of bulk density was substantially different than the value used in the TT model. ATSDR used a bulk density value of 1.65 g/cm^3 for Hadnot Point and Holcomb Boulevard. This is significantly lower than the 2.7 g/cm^3 used in the Tarawa Terrace model.

Additionally, ATSDR selected a K_d value of 0.30 mL/g[273] for PCE, which was different than the Tarawa Terrace value of 0.14 mL/g, even though the same soil characteristics are encountered in both areas.[274] Using these values for bulk density and K_d, the resulting retardation factor for PCE in Hadnot Point was 3.5. This value is much higher than the retardation factor of 2.93 that ATSDR used in the Tarawa Terrace model. Recall that a higher retardation factor means slower contaminant migration in groundwater.

In Summary (Opinion 14): ATSDR relied on very limited data for calibrating its fate and transport model for Hadnot Point. ATSDR admitted that they could not perform a rigorous evaluation of the model calibration, due to insufficient water-quality data. ATSDR made several arbitrary assumptions for assigning source-release locations, timing, and mass loading, to fit the limited water-quality data. ATSDR assumed constant mass loading of the same magnitude at all sources for more than 40 years, which is highly uncertain, if not impossible. This is indicative of the type of assumptions that were made by ATSDR for constructing its historical reconstruction model. ATSDR assigned model parameter values that were not based on site-specific data and were inconsistent with the values they used for the Tarawa Terrace model.

4.2.3.3 HP-634 Concentration Data were Incorrectly Interpreted

ATSDR's interpretation of the available sampling data at well HP-634 is incorrect. When reviewing the sampling data and the operation of well HP-634 during the period encompassing these sampling events, it is unlikely that this well was ever contaminated with elevated TCE concentrations, as ATSDR assumed.

A sample collected from well HP-634 on December 4, 1984, while the well was in operation, was a non-detection. The well ceased operations on December 6, 1984.[275] Another sample collected on December 10, 1984 was a non-detection.[276] However, a sample from January 16, 1985, when the well had been out of service for more than a month, had a reported concentration of 1,300 µg/L.[276] Two more samples collected in 1986 and 1991 were also non-detections.[276]

ATSDR assigned two sources in the model near well HP-634.[277] However, well HP-634 is located upgradient of these source locations and, therefore, contamination could not have reached that well when it was not operational. But even when the well was operational, the December 4, 1984, sample was a non-detection. Therefore, the reported concentration of 1,300 µg/L from the sample collected on January 16, 1984, when the well was not in service, should be considered erroneous. See Dr. Hennet's expert report for a more detailed discussion of this issue.[278]

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[272] ATSDR-HP, Chapter A, Supplement 6, p. S6.14

[273] ATSDR-HP, Chapter A, Supplement 6, p. S6.14

[274] See geohydrologic section A-A' in Hadnot Point-Holcomb Boulevard study area (ATSDR-HP, Chapter B, p. B12

[275] CLW0000006590

[276] ATSDR-HP, Chapter A, Table A4, p. A21

[277] Sources 1 & 2 in the HPIA

[278] Hennet (2024), Expert Report

However, ATSDR ignored the non-detections at HP-634 during model calibration and assigned a constant TCE concentration of 640,000 µg/L in layers 1, 2 and 3 for Sources 1 and 2, so that the historical reconstruction of concentrations at HP-634 would be close to the highest value reported (1,300 µg/L). Based on the rationale presented above, the source strength at those locations is substantially exaggerated and not supported by data. See more details in the discussion provided in Dr. Hennet's expert report.

If, instead, a lower mass loading at Sources 1 and 2 were considered, reconstruction of TCE concentrations at well HP-634 and the influent to the HP WTP would be lower. Such an adjustment would not fully eliminate the source in that area. The source adjustment would acknowledge some uncertainty in the sampling data and history of contamination at that well, given the findings in the soils near ground surface (see Hennet, 2024). For example, reducing the mass loading to 10% of the value specified in the ATSDR model, model simulated concentrations would be only a fraction of those calculated by ATSDR at well HP-634, but much closer to the non-detections observed at that well. This would also result in lower concentrations at influent to the treatment plant over the period 1962-1972. Figure 31 depicts the historical reconstruction of TCE concentrations at well HP-634 when such a reduction in mass loading is implemented.[279] The historical reconstruction of TCE concentrations in the influent to the treatment plant is depicted in Figure 32.

However, the Hadnot Point model is riddled with uncertainties and, therefore, the correction implemented in the model should not be considered as the only possible adjustment to the model inputs. These graphical representations serve only to demonstrate how variable the model outputs are to changes in parameters. They should not be interpreted as definitive, mean monthly concentrations of contaminants in the influent to the WTP.

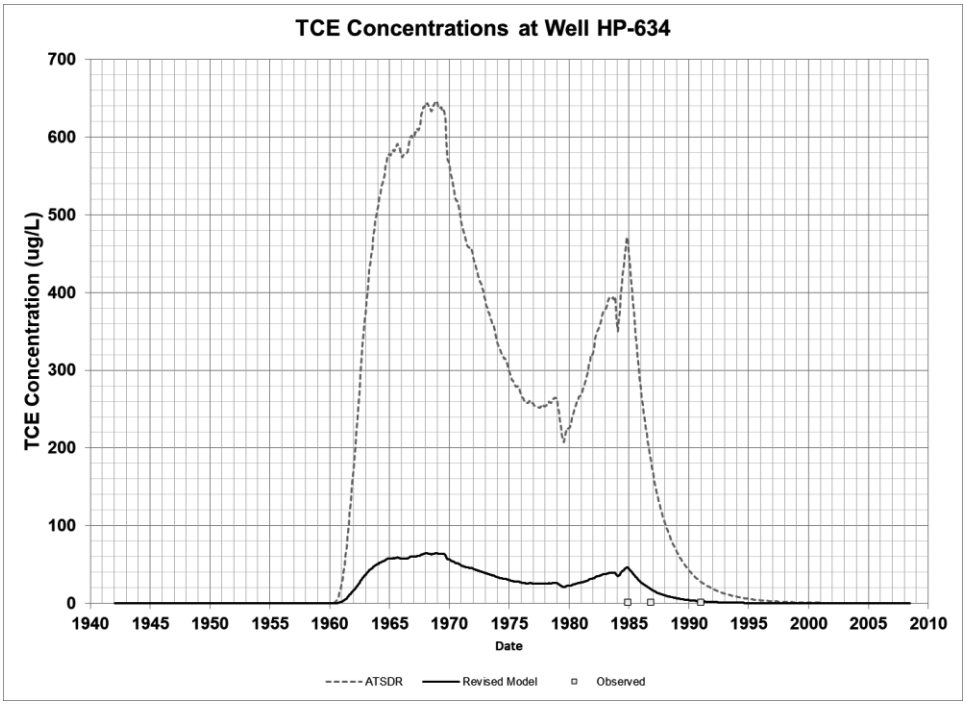

Figure 31: Changes in Model Simulated TCE Concentrations at Water-Supply Well HP-634 After Source Mass-Loading Correction

[279] The erroneous concentration of 1,300 µg/L is not depicted in this plot.

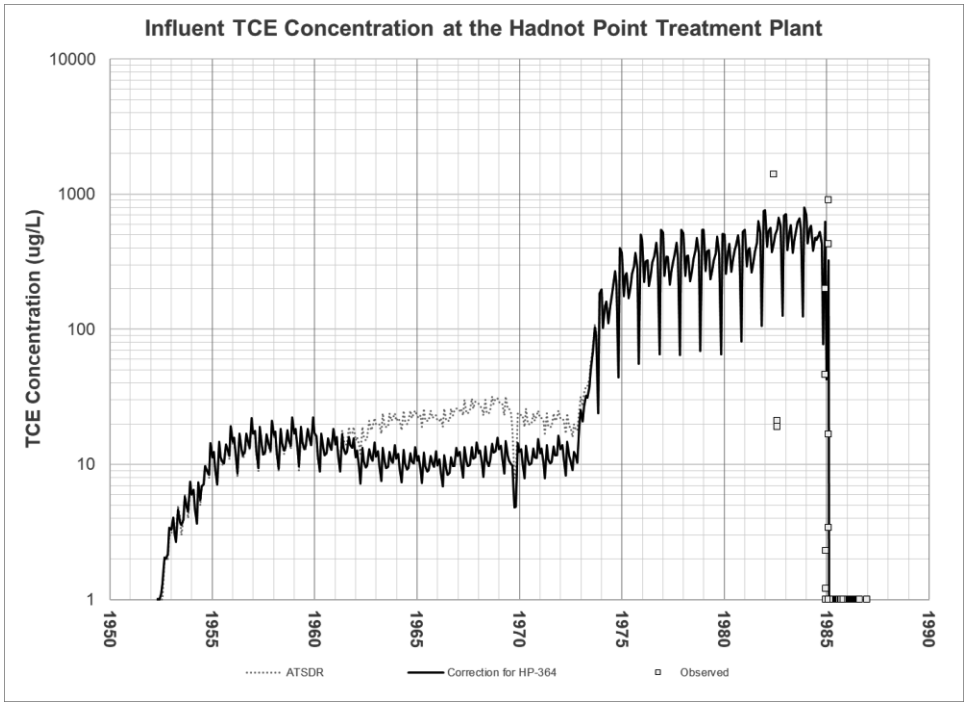

Figure 32: Changes in Model Simulated Historical Reconstruction of TCE Concentrations in the Influent to the HP WTP after Source Mass-Loading Correction

In Summary (Opinion 14 and 15): ATSDR incorrectly interpreted field sampling data. ATSDR focused on an elevated, but likely erroneous, concentration value, and ignored multiple non-detections reported for that well. Hence, ATSDR assumed presence of contamination in an area where the available data indicated either the absence or presence of only traces of contamination. ATSDR included the erroneous, elevated concentration value in its model calibration and ignored the non-detections, resulting in conservative and biased-high simulated concentrations, not representative of aquifer conditions.

4.2.4 VOC Degradation By-Products

ATSDR used the Linear Control Model (LCM), an alternative methodology for reconstructing the historical concentrations of the VOC degradation by-products. Unlike MODFLOW and MT3DMS, the LCM does not simulate the physical and chemical processes in the aquifer. As ATSDR indicated, the LCM "*does not require site-specific knowledge of the spatial distribution of aquifer and transport properties (e.g., hydraulic conductivity, porosity, contaminant source concentration).*"[280] Instead, the method "*characterizes the aquifer, contaminant sources, and the dynamics of contaminant migration as a 'black box.*"[281]

ATSDR applied this methodology to reconstruct concentrations at well HP-651 and then used a simple mixing model to calculate the blended concentration at the HP WTP. Application of this methodology relied on the same limited set of observed data, available after 1985. As illustrated in Figure 33, the historical reconstruction prior to 1985 cannot be verified, due to lack of observed data for the period.

[280] ATSDR-HP, Chapter A, p. A37

[281] ATSDR-HP, Chapter A, p. A37

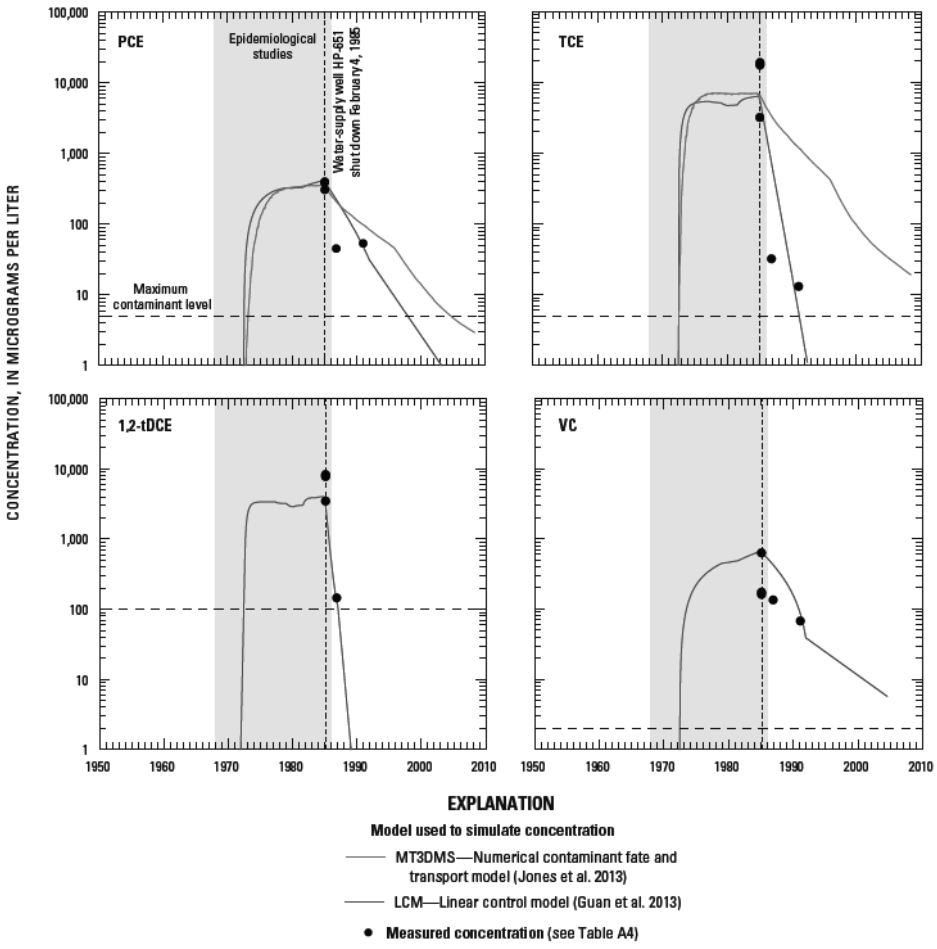

Figure A25. Reconstructed (simulated) concentrations of tetrachloroethylene (PCE), trichloroethylene (TCE), *trans*-1,2-dichloroethylene (1,2-tDCE), and vinyl chloride (VC) at water-supply well HP-651 using numerical (MT3DMS) and linear control methodology (TechControl) models, Hadnot Point–Holcomb Boulevard study area, U.S. Marine Corps Base Camp Lejeune, North Carolina. (See Figure A14 for well location.)

Figure 33: ATSDR's Model Simulated Concentrations of PCE, TCE, and their Degradation By-Products at Well HP-651

In Summary (Opinion 16): ATSDR used a "black box" for the historical reconstruction of VOC degradation by-products in Hadnot Point. However, the simulated historical reconstruction was not calibrated to historical data, as such data were not available prior to December 1984. Therefore, the estimated monthly contaminant concentrations cannot be verified.

4.2.5 Sensitivity and Uncertainty Analyses

ATSDR conducted a sensitivity analysis to evaluate the effect of variation of model input parameter values on model outputs. According to ATSDR, "*the results from all sensitivity analyses were used to define a range of finished-water concentrations at the HPWTP*." [282]

ATSDR also performed an uncertainty analysis of limited scope and magnitude, focusing on the "*effect of uncertainty in the pumping schedules of water-supply wells.*"[283]

Important aspects and conclusions of the sensitivity and uncertainty analyses are discussed below.

4.2.5.1 Sensitivity Analysis Results are Incorrectly Presented as Uncertainty Ranges

A discussion on key aspects of the sensitivity analysis conducted by ATSDR is provided in the sections below.

4.2.5.1.1 The Start Date of Source Releases is Highly Uncertain

As mentioned above, ATSDR admitted that "*specific data pertinent to the timing of initial deposition of contaminants to the ground or subsurface, chronologies of waste-disposal operations, such as dates and times when contaminants were deposited in the HPLF, or descriptions of the temporal variation of contaminant concentrations in the subsurface generally are not available.*"[284] In the absence of this critical information, reconstruction of historical conditions is highly uncertain. The timing and magnitude of contaminant releases in the soils at Camp Lejeune, their downward migration to the aquifers below, and ultimately their travel in groundwater before reaching the pumping wells, are unknown.

One type of contaminant source that ATSDR considered in the Hadnot Point model is underground storage tanks (USTs). Historical records for the start date of releases from UST systems were not available.[285] ATSDR formulated a rationale for delineating the start date of such releases from USTs, considering probable installation dates and data from a U.S. Environmental Protection Agency (EPA) study of UST releases.[286] The EPA report published in 1986 included an analysis of 12,444 leak incident reports across the United States.[287] These results indicated that the mean and median time of UST leaks was 11 and 9 years, respectively.[286] ATSDR used the median value of 9 years from the assumed installation date of the USTs to assign contaminant release start dates in the calibrated model.[286] The empirical data for UST releases may or may not be applicable to the USTs installed at Camp Lejeune and, therefore, assignment of timing and magnitude for these sources is arbitrary and uncertain.

ATSDR selected a source-release timeframe of 7 years for the landfill area based on the following rationale: "*given the lack of historical information, a similar source-release time frame, in this case 7 years from site development, was applied to HPLF-area sources within the model. The shorter source-release time frame acknowledges that landfill disposal likely encompassed a range of contained and uncontained source materials, in contrast to the engineered tank and piping system sources discussed previously*."[288]

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[282] ATSDR-HP, Chapter A, Supplement 6, p. S6.32

[283] ATSDR-HP, Chapter A, Supplement 6, p. S6.44

[284] ATSDR-HP, Chapter A, Supplement 6, p. S6.45

[285] ATSDR-HP, Chapter A, Supplement 6, p. S6.16

[286] ATSDR-HP, Chapter A, Supplement 6, p. S6.16

[287] The number of leak incident reports analyzed by USEPA (1,244) and the associated referenced document are incorrectly stated in ATSDR's report (Chapter A, Supplement 6, p. S6.16). The correct number is 12,444 (USEPA, 1986, "*Summary of State Reports on Releases from Underground Storage Tanks*," EPA/600/M-86/020, August 1986)

[288] ATSDR-HP, Chapter A, Supplement 6, p. S6.42

This statement further reinforces the argument regarding the arbitrary and uncertain timing and magnitude of contaminant source releases implemented in the model.

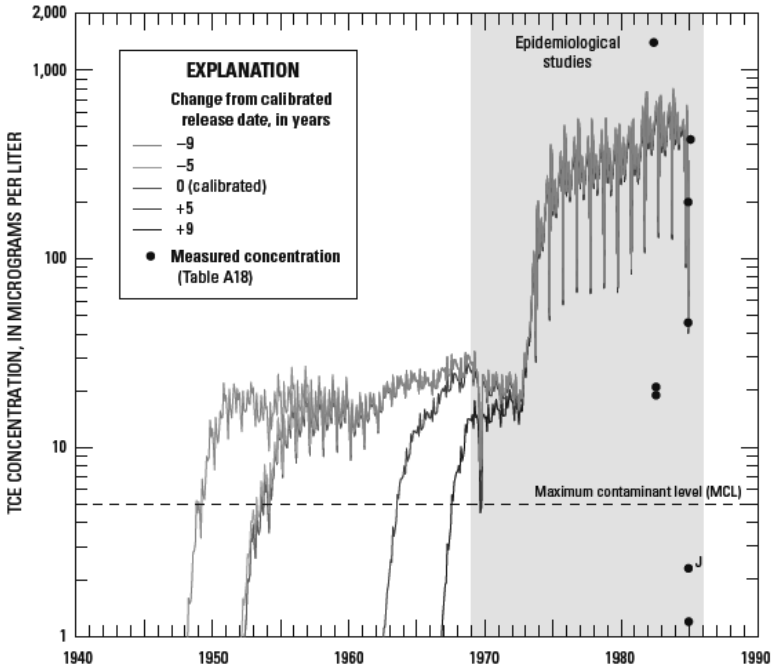
ATSDR proceeded to assess the model sensitivity to source-release dates by simulating scenarios where the start date was modified by ±5 and ±9 years.[289] This means ATSDR compared the calibrated model to models where the contamination release at each source began:

- 9 years before the calibrated model;
- 5 years before the calibrated model;
- 5 years after the calibrated model; and
- 9 years after the calibrated model.

The results of the corresponding calculations are depicted in Figure 34. ATSDR's sensitivity analysis results for these scenarios demonstrated the impacts of the arbitrary and uncertain assignment of source release timeframe and magnitude. Depending on when ATSDR assumed source releases began, contamination could have arrived at the pumping wells as early as 1948 or as late as 1967. This is a very wide timeframe, over which groundwater may or may not have been contaminated. It is also important to recall that, according to ATSDR "*[t]he core period of interest for the epidemiological studies is 1968–1985.*"[238]

The timing and magnitude of contaminant releases to the soils in Camp Lejeune are some of the most important parameters in the historical reconstruction of contaminant concentrations in the influent to the WTP. Without this critical information, any calculation on the arrival of contamination to the pumping wells is highly uncertain**.** ATSDR's sensitivity analysis on the source-release start date illustrates that it is impossible to know when such releases occurred, and there are no data to confirm or reject any hypothesis on those dates. Hence, it is not possible to validate the results of the calibrated model or any of the other sensitivity or uncertainty analyses performed by ATSDR and, therefore, the historical reconstruction of contamination in HPHB is arbitrary and uncertain.

[289] ATSDR-HP, Chapter A, Supplement 6, p. S6.42



[1]Change from calibrated release date, in years	First month exceeding MCL	Concentration at start of epidemiological study (January 1968), in micrograms per liter	Maximum concentration during epidemiological study period (January 1968–December 1985), in micrograms per liter
–9	November 1948	26	800
–5	April 1953	26	798
0 (calibrated)	August 1953	27	783
+5	August 1963	23	748
+9	August 1967	7	740

[1]Calibrated release date varies by source location (Table A13)

Note:
–9 years means 9 years earlier than calibrated-source release date
+9 years means 9 years after calibrated-source release date

Figure A37. Reconstructed (simulated) finished-water concentrations of trichloroethylene (TCE) derived from variations in contaminant-source release dates, Hadnot Point water treatment plant, Hadnot Point–Holcomb Boulevard study area, U.S. Marine Corps Base Camp Lejeune, North Carolina. [J, estimated]

Figure 34: Results of Sensitivity Analysis on Source Release Start Date

In Summary (Opinion 17): ATSDR conducted a sensitivity analysis to assess the impact of different source-release start dates on groundwater contamination. This is because there are no observed data about the actual time when USTs or other sources started to release contamination in the soil. As a result, it is not possible to confidently determine the actual period of groundwater contamination and, therefore, ATSDR's historical reconstruction is highly uncertain.

4.2.5.1.2 Sensitivity Analysis Scenarios Considered Extreme Parameter Values, and the Biased-High Results were Incorrectly Presented as Uncertainty Bounds

ATSDR constructed three scenarios to explore parameter sensitivity for the historical reconstruction of PCE and TCE concentrations.[290] In each scenario, the values of select hydraulic, fate, and transport model parameters were varied, and the resulting PCE and TCE concentrations in the influent to the WTP were calculated from the corresponding model results. The scenarios that ATSDR evaluated were the following:

- Scenario 1: Kxx, C, n, and α_L
- Scenario 2: Kxx, I, C, K_d, and λ
- Scenario 3: Kxx, I, C, n, α_L, K_d, and λ

where:

Kxx: horizontal hydraulic conductivity in Layer 3 (the most sensitive layer to hydraulic conductivity variability)

I: infiltration (areal recharge)

C: contaminant source concentration

n: porosity (effective; not total)

a_L: longitudinal dispersivity

K_d: distribution coefficient

λ: biodegradation rate

In these scenarios, ATSDR conducted an ensemble of model simulations that were part of a sensitivity analysis but essentially resembled a form of uncertainty analysis. Unlike a sensitivity analysis, which explores model response to parameter variability, uncertainty analysis is used to construct a range of possible outcomes. ATSDR claimed that they performed a sensitivity analysis, but presented the results as if it provided an uncertainty range. As will be illustrated below, the resulting range is not a reliable estimate of the possible uncertainty of model results, as the assumptions underpinning this analysis are not consistent with assumptions built into the calibrated flow and transport model. In fact, by doing so ATSDR conflated the sensitivity and uncertainty analyses.

In order to construct these sensitivity scenarios, ATSDR selected, for each parameter, the two extremes of the range of values for this parameter and ran the model twice, one using the minimum value of the parameter and one with the maximum value. For example, the two values selected for the hydraulic conductivity in Layer 3 were equal to 0.1 and 10 times the calibrated value.[291] Thus, if the horizontal hydraulic conductivity in the HPIA varies between 1 to 50 ft/d,[292] the values used in the sensitivity scenario would be as low as 0.1 ft/d, or as high as 500 ft/d.

To understand the importance of this assumption, recall that, for the Tarawa Terrace uncertainty analysis, ATSDR defined reasonable ranges for the calibrated parameter values. Regarding the variability in hydraulic conductivity, uncertainty realizations with hydraulic conductivity values that would exceed an acceptable range of model calibration were excluded from the analysis. No such caution was applied in the sensitivity analysis for Hadnot Point and Holcomb Boulevard. The values used in the HPHB sensitivity

[290] ATSDR-HP, Chapter A, p. A79

[291] ATSDR-HP, Chapter A, p. A79

[292] ATSDR-HP, Chapter A, p. A41

analysis represented two extremes of the range. Recall that for the uncertainty analysis in Tarawa Terrace, ATSDR implemented filtering criteria to reject implausible realizations that would violate calibration criteria for the groundwater flow model. In the HPHB sensitivity analysis such criteria were not implemented.

The implication of this choice is ATSDR used the calibrated model for this analysis, but then they varied parameter values to extremes, without evaluating how that would impact model calibration. Thus, either the model was reasonably calibrated (and, therefore, such wide range of parameter values was not warranted), or the model was not reasonably calibrated (and, therefore, a wide range of parameter values was warranted).

This issue is further exacerbated by ATSDR's use of extreme values for fate and transport simulations in Hadnot Point. The discussion below illustrates the implications of this approach.

Recall that in the Tarawa Terrace model, ATSDR defined a range of values for transport parameters based on literature sources and professional judgment. ATSDR proceeded with defining probabilistic distributions of these parameters, to calculate parameter value ranges for the uncertainty analysis (Section 4.1.2.2). The probabilistic distributions forced parameter ranges to vary closely around the mean value of the parameter, which was determined during model calibration.

For the Hadnot Point model, ATSDR did not conduct such an analysis for defining appropriate parameter ranges. Instead, ATSDR selected extreme values for the fate and transport parameters, corresponding to the 2.5 and 97.5 percentile of the parameter range. Table 2 lists the parameter values that ATSDR used for calculating the retardation factor for their sensitivity analysis.

It is important to understand the impact of selecting such extreme parameter values, and especially the maxima of these ranges. For example, a porosity value of 0.11 means that groundwater velocities can be double the mean, calibrated value. Recall that for the Tarawa Terrace analysis (Section 4.1.3.2), ATSDR used a porosity value that, on average, was equal to the calibrated value of 0.2. In other words, in Tarawa Terrace the uncertainty analysis included 840 realizations for which porosity was, on average, unchanged from the calibrated value. However, in Hadnot Point, the sensitivity analysis was presented in lieu of an uncertainty analysis, and calculations used values 50% higher or lower than those ATSDR deemed reasonable for the Tarawa Terrace uncertainty analysis.

Table 2. ATSDR's Sensitivity Analysis Range of Parameter Values for Calculating the Retardation Factor for PCE and TCE in Hadnot Point[a]

Contaminant	Statistic	K_d (mL/g)[b]	Porosity	Bulk Density (g/cm^3)[c]
PCE	Mean	0.31	0.20	1.66
	Minimum (2.5 Percentile)	**0.15**	**0.11**	1.62
	Maximum (97.5 Percentile)	**0.45**	**0.29**	1.70
TCE	Mean	0.15	0.2	1.66
	Minimum (2.5 Percentile)	**0.08**	**0.11**	1.62
	Maximum (97.5 Percentile)	**0.22**	**0.29**	1.70

[a] Values from ATSDR's Table S6.8 (ATSDR-HP, Chapter A, Supplement 6, p. S6.35)

[b] Value converted from ft^3/g

[c] Value converted from g/ft^3

By considering such a wide range of parameter values and simulating groundwater flow and contaminant transport using extreme parameter values, the results of the sensitivity analysis would

unsurprisingly be wide, but not reasonable. Certain combinations of parameter values would include those extreme values that would result in the highest groundwater velocities (maximum hydraulic conductivity), highest contaminant source (maximum source concentration), and fastest contaminant migration (minimum K_d, minimum porosity, minimum biodegradation rate). Such combinations would result in the earliest arrivals and highest concentrations of the ranges shown in Figure 35. However, the minima and maxima for these parameters are not within a reasonable range that is consistent with site-specific data, or even the calibrated model parameter values. Hence, the calculated concentration ranges are not indicative of the potential variability of contaminant concentrations in the influent to the WTP.

It is also important to highlight that in the uncertainty analysis for Tarawa Terrace, ATSDR discarded "*physically implausible realizations*" for the groundwater flow model (Section 4.1.3.2), i.e., did not violate acceptance criteria for fitting observed water-level data. By doing so, ATSDR acknowledged the importance of generating realizations that would "calibrate" the model and replicating observed conditions. In the analysis for Hadnot Point such criteria for determining implausible realizations were not applied.

Nonetheless, Mr. Maslia said in his expert report: "*Based on these results, it is scientifically defensible to conclude that during the period of the 1950s to the mid-1980s, contaminant concentration levels would have occurred within this range of values (the shaded region) at HPWTP, with the average (most likely) values being the solid line in the interior*."[293]

In Summary (Opinion 18): ATSDR conducted a sensitivity analysis of their calibrated model by selecting extreme values for model parameters. These extreme values were not supported by site-specific data. Model simulations were performed for combinations of these extreme values. Paticular combinations of these extreme parameter values resulted in conservative and biased-high estimates of monthly concentrations. Although a sensitivity analysis is designed to determine the impact of parameter value changes to model outcomes, ATSDR presented the results of this analysis as indicative of the expected range of reconstructed monthly contaminant concentrations. This is not correct.

[293] Maslia (2024), Expert Report, p.88

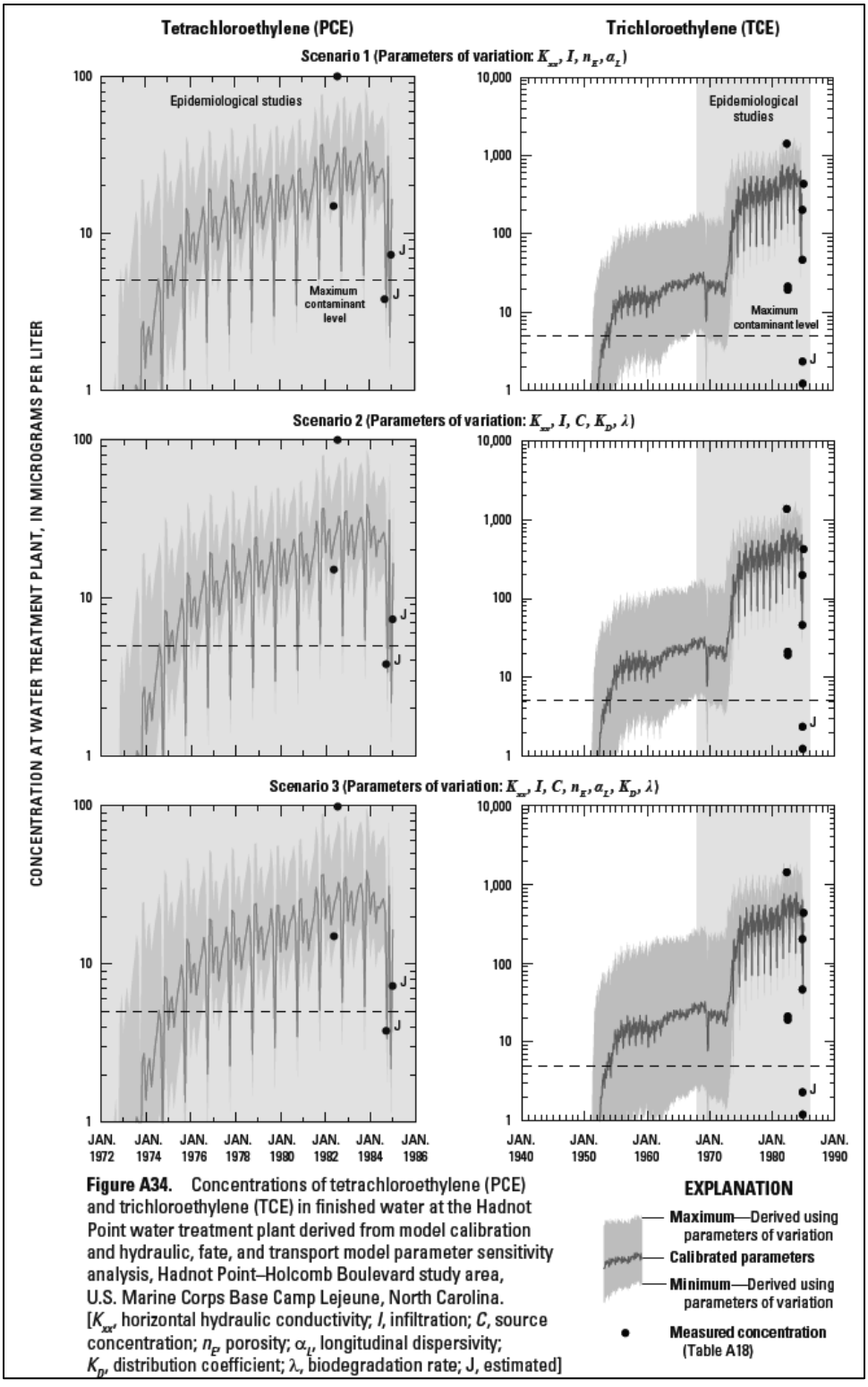


Figure A34. Concentrations of tetrachloroethylene (PCE) and trichloroethylene (TCE) in finished water at the Hadnot Point water treatment plant derived from model calibration and hydraulic, fate, and transport model parameter sensitivity analysis, Hadnot Point–Holcomb Boulevard study area, U.S. Marine Corps Base Camp Lejeune, North Carolina. [K_{xx}, horizontal hydraulic conductivity; I, infiltration; C, source concentration; n_E, porosity; α_L, longitudinal dispersivity; K_D, distribution coefficient; λ, biodegradation rate; J, estimated]

Figure 35: ATSDR's Sensitivity Analysis of PCE and TCE Concentrations in the Influent to the HPHB WTP

4.2.5.2 Uncertainty Analysis Is Incomplete

The uncertainty analysis conducted by ATSDR focused solely on the uncertainty of the pumping schedules of water supply wells.[283] ATSDR implemented a Latin Hypercube Sampling (LHS) methodology, an approach that generates a limited number of scenarios to be evaluated. The reason for this choice is that a rigorous uncertainty analysis using, for example, a Monte Carlo approach, would require tens of thousands of realizations.[283] Such an approach would be computationally infeasible and, therefore, the simplified approach using LHS was selected instead.

However, ATSDR considered a small number of only 10 uncertainty scenarios. For each scenario of historical pumping schedule, a new input to the calibrated model was constructed to incorporate this pumping schedule, and the model was run to calculate the historical concentrations in the influent to the WTP. The simulation results of these scenarios were aggregated to the plot shown in Figure 36.

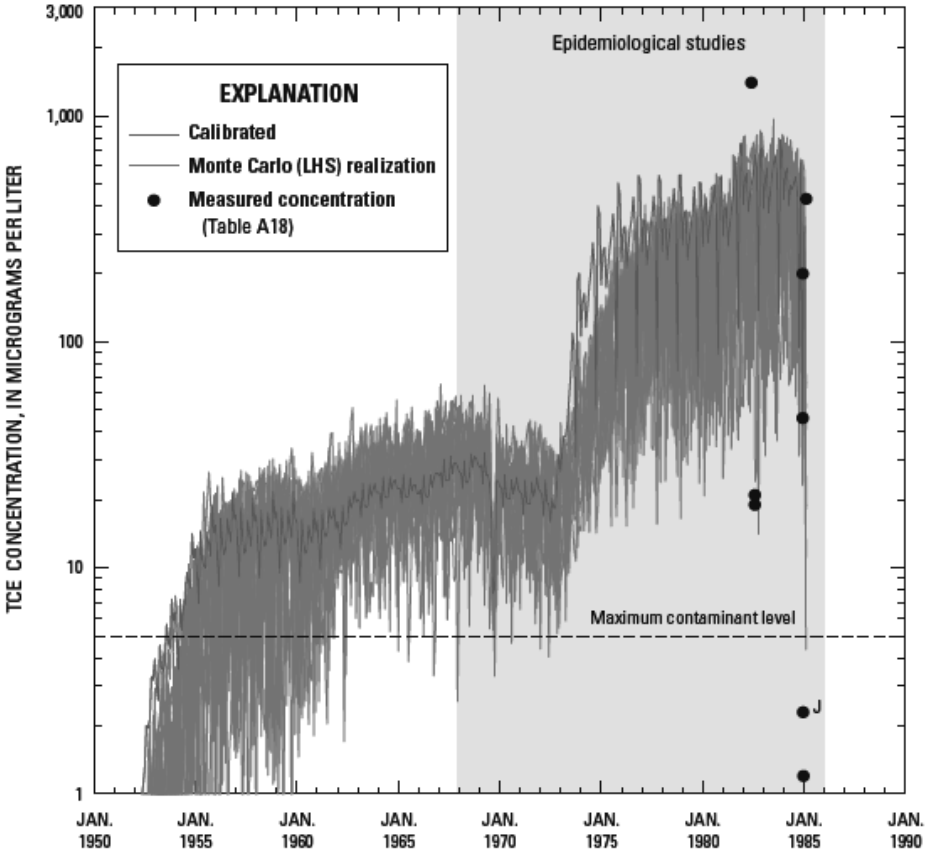


Figure S6.23. Variations in reconstructed (simulated) finished-water concentrations of trichloroethylene (TCE) derived using Latin hypercube sampling (LHS) methodology on water-supply well monthly operational schedules, Hadnot Point water treatment plant, Hadnot Point–Holcomb Boulevard study area, U.S. Marine Corps Base Camp Lejeune, North Carolina. [J, estimated]

Figure 36: TCE Variation due to Pumping Schedule Uncertainty

Recall the discussion in Section 3.1.5 about the general rule for the calibrated model output (prediction): "*[i]deally, the value of that prediction should lie somewhere near the centre of the uncertainty band of the prediction. In this way, the potential for predictive error is minimized*."[294] Inspection of Figure 36 indicates that the calibrated model fails to conform with this rule at two critical times: (a) in the early 1950s, when the model estimates the arrival of TCE at the pumping wells and, thus, the influent to the WTP, and (b) after 1972, when pumping well HP-651 was put in service.

In both cases, the calibrated model is at or above the upper bound of the uncertainty range calculated by ATSDR. This clearly indicates that, with respect to pumping variability, the contaminant migration timeframe calculated by ATSDR's calibrated model is biased high. The model estimates faster arrival of contaminants to the pumping wells and, therefore, the WTP, than the uncertainty analysis indicates.

Figure 36 also indicates that the reconstructed historical concentrations calculated by ATSDR's calibrated model are biased high, as they are not near the center of the uncertainty range but, instead, at or near the upper bound of the uncertainty range.

ATSDR's analysis only partially addressed model uncertainty.The upper and lower bounds of uncertainty, shown in Figure 36, represent 95% of Monte Carlo simulations for evaluating only the effects of pumping schedule uncertainty,[295] are therefore the resulting range of uncertainty is unreliable.

In Summary (Opinion 19): ATSDR conducted a limited uncertainty analysis, focusing solely on the effects of historical pumping variability on estimated monthly contaminant concentrations. This analysis only partially addressed the model uncertainty. Even this limited uncertainty analysis indicated that the calibrated model is conservative and biased high, as it is either at or above the upper bound of the calculated uncertainty range.

4.2.5.3 Concluding Remarks

As discussed above, the inherent shortcomings of ATSDR's model calibration, sensitivity analysis, and uncertainty analysis, limit the validity of the associated conclusions that ATSDR attempted to draw regarding the history of groundwater contamination in HPHB. The lack of site-specific data to confidently assign model parameters within reasonable ranges prevents a reliable model calibration. This is further exacerbated by the lack of historical observation data to constrain the calibration and ensure that historical patterns of contamination are reliably reproduced by the model. As Sepulveda and Doherty (2015) stated, the "*model should replicate observed system behavior."* The ATSDR model results did not meet this requirement. Similarly to Tarawa Terrace, there is no "observed system behavior" (i.e. historical data from the entire period of interest) to support a reasonable and accurate model calibration and, therefore, an accurate historical reconstruction of contaminant concentrations in the influent to the WTP.

But even the sensitivity and uncertainty analyses presented by ATSDR failed to quantify the uncertainty range reliably. In its own admission, ATSDR stated that *"the ranges of values presented in the sensitivity analysis section of this report assess a limited number of input and output model parameters. The results (i.e., range of concentration) presented in the sensitivity analysis reported herein should not be considered or interpreted as the results of a robust and comprehensive uncertainty analysis, but do provide insight into parameter sensitivity and uncertainty in a qualitative sense*."[296] This contradicts Mr. Maslia's statement about the results of the sensitivity analysis in his expert report, which was that "*it is scientifically defensible to conclude that during the period of the 1950s to the mid-1980s, contaminant concentration levels would have occurred within this range of values (the shaded region) at HPWTP, with the average*

[294] Doherty (2015), p. 52

[295] 2.5 and 97.5 percentile.

[296] ATSDR-HP, Chapter A, Supplement 6, p. S6.45

(most likely) values being the solid line in the interior."[293] If parameter sensitivity and uncertainty can only be evaluated in a qualitative sense, it is not scientifically defensible to conclude that contaminant concentration levels would have been within the range indicated by the sensitivity analysis performed by ATSDR. It is also not scientifically defensible to conclude that the most likely values of contamination were those calculated by the calibrated model.

Section 5
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APPENDICES

Appendix A: Site-Specific Data for K_d

Table A-1. Site-Specific Data for K_d

Sample	Date Sampled	Depth (ft)	TOC (mg/kg)	f_{oc}	K_{oc}	K_d (mL/g) All Values	K_d (mL/g) Final	Comments	Reference Citation
BLDG902-SB03-100-101-07B	5/20/2007	100.5	28,000	0.028000	234.42	6.5638	6.5638		CLJA_WATERMODELING_07-0001380126
BLDG902-SB03-10-11-07B	5/19/2007	10.5	810	0.000810	234.42	0.1899	0.1899		CLJA_WATERMODELING_07-0001380120
BLDG902-SB03-120-121-07B	5/20/2007	120.5	2,600	0.002600	234.42	0.6095	0.6095		CLJA_WATERMODELING_07-0001380127
BLDG902-SB03-25-26-07B	5/19/2007	25.5	210	0.000210	234.42	0.0492	0.0492		CLJA_WATERMODELING_07-0001380121
BLDG902-SB03-43-44-07B	5/20/2007	43.5	300	0.000300	234.42	0.0703	0.0703		CLJA_WATERMODELING_07-0001380122
BLDG902-SB03-46-47-07B	5/20/2007	46.5	24,000	0.024000	234.42	5.6261	5.6261		CLJA_WATERMODELING_07-0001380123
BLDG902-SB03-55-56-07B	5/20/2007	55.5	1,300	0.001300	234.42	0.3047	0.3047		CLJA_WATERMODELING_07-0001380124
BLDG902-SB03-83-84-07B	5/20/2007	83.5	1,200	0.001200	234.42	0.2813	0.2813		CLJA_WATERMODELING_07-0001380125
IR84-DP27	7/19/2001	0.5	13	0.000013	234.42	0.0030	0.0030		CLJA_WATERMODELING_05-0000408229; CLJA_WATERMODELING_05-0000408591; CLJA_WATERMODELING_07-0001379092
IR84-DP36	7/20/2001	0.5	3	0.000003	234.42	0.0007		Excluded; inconsistent with the two other samples	CLJA_WATERMODELING_05-0000408232; CLJA_WATERMODELING_05-0000408600; CLJA_WATERMODELING_07-0001379092

Table A-1. Site-Specific Data for K_d

Sample	Date Sampled	Depth (ft)	TOC (mg/kg)	f_{oc}	K_{oc}	K_d (mL/g) All Values	K_d (mL/g) Final	Comments	Reference Citation
								from the same date	
IR84-DP36	7/20/2001	0.5	220	0.000220	234.42	0.0516	0.0527	Average of two samples	CLJA_WATERMODELING_07-0001379092
IR84-DP36	7/20/2001	0.5	230	0.000230	234.42	0.0539			CLJA_WATERMODELING_07-0001379092
IS26-04	11/21/1997	16.5	1,510	0.001510	234.42	0.3540	0.3540		CLJA_WATERMODELING_01-0000283421; CLJA_WATERMODELING_01-0000283606
IS26-05	11/21/1997	18	5,560	0.005560	234.42	1.3034	1.3034		CLJA_WATERMODELING_01-0000283421; CLJA_WATERMODELING_01-0000283607
IS26-06	11/21/1997	19	6,420	0.006420	234.42	1.5050	1.5050		CLJA_WATERMODELING_01-0000283421; CLJA_WATERMODELING_01-0000283608
SWMU253-TW02	3/22/2002	10	2,005	0.002005	234.42	0.4700	0.4700		CLJA_WATERMODELING_07-0002047135; CLJA_WATERMODELING_07-0002045499; CLJA_WATERMODELING_07-0001379091
SWMU254-SS01	7/18/2000	10	2,500	0.002500	234.42	0.5861	0.7173	Average of two samples	CLJA_WATERMODELING_01-0000259216; CLJA_WATERMODELING_01-0000259590; CLJA_WATERMODELING_07-0001379091
SWMU254-SS01	7/18/2000	10	3,620	0.003620	234.42	0.8486			CLJA_WATERMODELING_01-0000259216;

Table A-1. Site-Specific Data for K_d

Sample	Date Sampled	Depth (ft)	TOC (mg/kg)	f_{oc}	K_{oc}	K_d (mL/g) All Values	K_d (mL/g) Final	Comments	Reference Citation
									CLJA_WATERMODELING_01-0000259588; CLJA_WATERMODELING_07-0001379091
SWMU254-TW02	3/22/2002	8	2,084	0.002084	234.42	0.4885	0.4885		CLJA_WATERMODELING_07-0002047135; CLJA_WATERMODELING_07-0002045509; CLJA_WATERMODELING_07-0001379091
SWMU255-GW01	3/21/2002	8	824	0.000824	234.42	0.1932	0.1932		CLJA_WATERMODELING_07-0002047135; CLJA_WATERMODELING_07-0002045522; CLJA_WATERMODELING_07-0001379091
SWMU255-SS01	7/18/2000	8	444	0.000444	234.42	0.1041	0.1041		CLJA_WATERMODELING_01-0000259216; CLJA_WATERMODELING_01-0000259591; CLJA_WATERMODELING_07-0001379091
SWMU256-GW02	7/18/2000	2	2,470	0.002470	234.42	0.5790	0.5790		CLJA_WATERMODELING_01-0000259216; CLJA_WATERMODELING_01-0000259592; CLJA_WATERMODELING_07-0001379091
SWMU256-GW03	7/18/2000	2	3,460	0.003460	234.42	0.8111	0.8111		CLJA_WATERMODELING_01-0000259216; CLJA_WATERMODELING_01-0000259593;

Table A-1. Site-Specific Data for K_d

Sample	Date Sampled	Depth (ft)	TOC (mg/kg)	f_{oc}	K_{oc}	K_d (mL/g) All Values	K_d (mL/g) Final	Comments	Reference Citation
									CLJA_WATERMODELING_07-0001379091
SWMU258-GW01	3/22/2002	4	1,916	0.001916	234.42	0.4492	0.4492		CLJA_WATERMODELING_07-0002047135; CLJA_WATERMODELING_07-0002045538; CLJA_WATERMODELING_07-0001379091
SWMU258-GW02	7/18/2000	14	30,400	0.030400	234.42	7.1265	7.1265		CLJA_WATERMODELING_01-0000259216; CLJA_WATERMODELING_07-0001379091
SWMU258-IS05	7/17/2000	2	1,820	0.001820	234.42	0.4266	0.4266		CLJA_WATERMODELING_01-0000259216; CLJA_WATERMODELING_01-0000259595; CLJA_WATERMODELING_07-0001379091
SWMU261-GW02	7/18/2000	14	3,930	0.003930	234.42	0.9213	0.9213		CLJA_WATERMODELING_01-0000259216; CLJA_WATERMODELING_01-0000259597; CLJA_WATERMODELING_07-0001379091
SWMU264-GW01	3/24/2002	8	4,167	0.004167	234.42	0.9768	0.9768		CLJA_WATERMODELING_07-0002047135; CLJA_WATERMODELING_07-0002045568; CLJA_WATERMODELING_07-0001379092
SWMU264-SS01	7/17/2000	8	578	0.000578	234.42	0.1355		Excluded; the two samples	CLJA_WATERMODELING_01-0000259216;

Table A-1. Site-Specific Data for K_d

Sample	Date Sampled	Depth (ft)	TOC (mg/kg)	f_{oc}	K_{oc}	K_d (mL/g) All Values	K_d (mL/g) Final	Comments	Reference Citation
								are inconsistent	CLJA_WATERMODELING_01-0000259599; CLJA_WATERMODELING_07-0001379092
SWMU264-SS01	7/17/2000	8	9,800	0.009800	234.42	2.2973			CLJA_WATERMODELING_01-0000259216; CLJA_WATERMODELING_01-0000259598; CLJA_WATERMODELING_07-0001379092
SWMU265-GW02	3/24/2002	10	976	0.000976	234.42	0.2288	0.2288		CLJA_WATERMODELING_07-0002047135; CLJA_WATERMODELING_07-0002045576; CLJA_WATERMODELING_07-0001379092
SWMU285-GW01	3/26/2002	4	11,350	0.011350	234.42	2.6607	2.6607		CLJA_WATERMODELING_07-0002047135; CLJA_WATERMODELING_07-0002045613; CLJA_WATERMODELING_07-0001379091
SWMU285-GW03	7/19/2000	8	784	0.000784	234.42	0.1838	0.1838		CLJA_WATERMODELING_01-0000259216; CLJA_WATERMODELING_01-0000259604; CLJA_WATERMODELING_07-0001379091
SWMU294-GW01	3/22/2002	8	1,559	0.001559	234.42	0.3655	0.3655		CLJA_WATERMODELING_07-0002047135; CLJA_WATERMODELING_07-0002045644;

Table A-1. Site-Specific Data for K_d

Sample	Date Sampled	Depth (ft)	TOC (mg/kg)	f_{oc}	K_{oc}	K_d (mL/g) All Values	K_d (mL/g) Final	Comments	Reference Citation
									CLJA_WATERMODELING_07-0001379091
SWMU295-GW01	3/22/2002	8	1,966	0.001966	234.42	0.4609	0.4609		CLJA_WATERMODELING_07-0002047135; CLJA_WATERMODELING_07-0002045680; CLJA_WATERMODELING_07-0001379091
SWMU311-GW03	3/25/2002	6	1,364	0.001364	234.42	0.3198	0.3198		CLJA_WATERMODELING_07-0002047135; CLJA_WATERMODELING_07-0002045826; CLJA_WATERMODELING_07-0001379091
SWMU312-GW01	3/21/2002	4	2,005	0.002005	234.42	0.4700	0.4700		CLJA_WATERMODELING_07-0002047135; CLJA_WATERMODELING_07-0002045840; CLJA_WATERMODELING_07-0001379091
SWMU360-TW04	3/25/2002	12	875	0.000875	234.42	0.2051	0.2051		CLJA_WATERMODELING_07-0002047135; CLJA_WATERMODELING_07-0002046015; CLJA_WATERMODELING_07-0001379091
SWMU361-TW01	3/24/2002	4	1,216	0.001216	234.42	0.2851	0.2851		CLJA_WATERMODELING_07-0002047135; CLJA_WATERMODELING_07-0002046030; CLJA_WATERMODELING_07-0001379091

Table A-1. Site-Specific Data for K_d

Sample	Date Sampled	Depth (ft)	TOC (mg/kg)	f_{oc}	K_{oc}	K_d (mL/g) All Values	K_d (mL/g) Final	Comments	Reference Citation
SWMU362-SB01	3/26/2002	2	13,670	0.013670	234.42	3.2046	3.2046		CLJA_WATERMODELING_07-0002047135; CLJA_WATERMODELING_07-0002046032; CLJA_WATERMODELING_07-0001379091
SWMU43-GW01	7/18/2000	14	589	0.000589	234.42	0.1381	0.1381		CLJA_WATERMODELING_01-0000259216; CLJA_WATERMODELING_01-0000259586; CLJA_WATERMODELING_07-0001379092
SWMU43-GW02	3/25/2002	12	719	0.000719	234.42	0.1686	0.1686		CLJA_WATERMODELING_01-0000259216; CLJA_WATERMODELING_01-0000259580; CLJA_WATERMODELING_07-0001379092
SWMU43-GW02	7/17/2000	14	341	0.000341	234.42	0.0799	0.0799		CLJA_WATERMODELING_07-0002047135; CLJA_WATERMODELING_07-0002045472; CLJA_WATERMODELING_07-0001379092
SWMU43-GW03	7/17/2000	14	239	0.000239	234.42	0.0560	0.0897	Average of two samples	CLJA_WATERMODELING_01-0000259216; CLJA_WATERMODELING_01-0000259582; CLJA_WATERMODELING_07-0001379092
SWMU43-GW03	7/18/2000	14	526	0.000526	234.42	0.1233			CLJA_WATERMODELING_07-0001379092

Table A-1. Site-Specific Data for K_d

Sample	Date Sampled	Depth (ft)	TOC (mg/kg)	f_{oc}	K_{oc}	K_d (mL/g) All Values	K_d (mL/g) Final	Comments	Reference Citation
SWMU53-GW01	3/21/2002	6	21,150	0.021150	234.42	4.9580	4.9580		CLJA_WATERMODELING_07-0002047135; CLJA_WATERMODELING_07-0002045493; CLJA_WATERMODELING_07-0001379091

Appendix B: Discussion on Site-Specific K_d Values

According to the published data referenced by ATSDR, the range of K_d values for sands was 0.25-0.76 mL/g, averaging 0.39 mL/g; the referenced range of values for silts was 0.21-0.71 mL/g, averaging 0.40 mL/g. ATSDR used the mean value of 0.40 mL/g as the starting value in the model calibration process. However, the value determined by ATSDR through model calibration was 0.14 mL/g. This value is out of range for the soils in the aquifer at Camp Lejeune, as discussed below.

In addition, this value is more than two times lower than the value used by ATSDR for the Hadnot Point model, despite ATSDR's statement that the same aquifers are encountered in both models.[297] ATSDR ultimately defined a K_d value of 0.30 mL/g for PCE, through "refinement during the model calibration process."[298]

Site-specific Total Organic Carbon (TOC) data were available when the Tarawa Terrace and Hadnot Point models were constructed, which ATSDR did not consider. Table A-1 in Appendix A lists available TOC data. These data provide site-specific estimates of fraction organic carbon (f_{oc}) which, multiplied by the chemical-specific K_{oc}, provide K_d estimates for the Tarawa Terrace aquifers.

Based on Table A-1 in Appendix A, site-specific estimates of fraction organic carbon (f_{oc}) can be calculated by dividing the TOC data by 10^{+6}, to express the values in percentages of mg/mg. The resulting f_{oc} values vary between 0.000003 and 0.0304. Corresponding K_d values are calculated using a PCE-specific K_{oc} value of 234.42.[299]

The K_d range calculated using site-specific data is consistent with the range provided by ATSDR, with an upper bound lower than the maximum of the ATSDR-cited range. However, the site-specific data provide a distribution of K_d values that reflects actual conditions in the Castle Hayne aquifer, and that can be used for calculating site-specific statistics for this parameter.

When duplicate samples are replaced by their average and extreme values are removed, the mean value is 1.0999 mL/g. The corresponding geometric mean and median values are 0.3998 and 0.3961 mL/g, respectively. When samples from depths greater than 10 ft are considered, the mean K_d value is actually higher (1.2858 mL/g). The corresponding geometric mean and median values are 0.4244 and 0.3047 mL/g, respectively.

These values are two to three times greater than the value used by ATSDR in the Tarawa Terrace model (0.14 mL/g). They are also greater than the value used by ATSDR in the Hadnot Point model (0.30 mL/g), except for the median value for samples from depths greater than 10 ft.

The implications of using a K_d that is substantially lower than the mean value derived from measured data are important, as the corresponding retardation coefficients can be significantly underestimated, resulting in faster plume migration and, therefore, erroneously shorter contaminant arrival times at the pumping wells.

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[297] "*Because field data describing contaminant fate and transport parameters is lacking for the HPHB study area and the TT study area is adjacent to the HPHB study area, the probability density functions described by Maslia et al. (2009) were used to generate a range of transport parameters values for the analyses reported herein*." (ATSDR-HP, Chapter A, Supplement 6, p. S6.34)

[298] ATSDR assumed an f_{oc} value of 0.002, and a range of K_{oc} values from the literature (ATSDR-HP, Chapter A, Supplement 6, Table S6.4, p. S6.14). In that same table, ATSDR provided the corresponding calculated range of K_d values for PCE, varying between 0.03 and 21.43 mL/g (the K_d value is the same when units of L/kg are used).

[299] Value derived from logKoc = 2.37 (ATSDR-TT, Chapter D, Table D12, p. D15)

Appendix C: Discussion on ATSDR's Retardation Factor Calculations

Below is a detailed discussion on: (a) ATSDR's probabilistic distributions and ranges for the parameters used for calculating the retardation factor, and (b) the resulting spatial distribution of K_d in the model cells and the resulting implication in the calculation of the retardation factor.

C-1. Probabilistic distributions and associated ranges

ATSDR conducted uncertainty analysis of the Tarawa Terrace historical reconstruction of contaminant concentrations in the finished water produced by the treatment plant. The uncertainty analysis assessed the range of model outputs due to variability in model parameter values. ATSDR defined model parameter uncertainty by constructing probabilistic distributions of those parameters, based on published ranges of the parameter values.

For the distribution coefficient (K_d) for PCE, ATSDR selected a range of values to consider in its uncertainty analysis, corresponding to a reasonable range for silts and sands from comparable published data.[300] A probability density function was developed using a mean and standard deviation. The K_d value used in the calibrated model ($5.0x10^{-6}$ ft^3/g, or 0.14 mL/g) was defined as the mean for the probabilistic distribution. The rationale for selecting the parameter standard deviation for generating the probability density function is not documented.

For one of the uncertainty scenarios, which excluded pumping uncertainty, ATSDR constructed 840 realizations. Using the selected mean and standard deviation values, ATSDR generated 840 K_d sets, with values assigned on a cell-by-cell basis for each groundwater model realization. Statistics of the PDF illustrated on the graph in Figure 37, calculated from files generated by ATSDR[301]. The horizontal axis shows K_d values in ft^3/g, and corresponding percentages of that value for the ensemble of the 840 distributions are shown on the vertical axis. A secondary horizontal axis at the top of the graph shows the same K_d values but different units (mL/g) used interchangeably in ATDR's reports. The blue horizontal line at the top of the graph depicts the range of values corresponding to ATSDR's minimum and maximum value (blue dots at the edges of the blue line) for the statistical description of the probabilistic distribution. The middle blue dot on that line indicated the selected mean value of the probability distribution. As illustrated in this graph, the actual distribution of K_d values that was generated by the algorithm when all 840 realizations are considered, spans only a fraction of the range that ATSDR indicated as representative for soils similar to those found at Camp Lejeune.

ATSDR's algorithm for generating random numbers was set to exclude values outside the prescribed range. However, based on the statistical description for K_d, only values below the prescribed minimum were omitted, as higher values never reached the upper bound of the range, further highlighting the reasons for the skewed range of values illustrated in Figure 37.

[300] ATSDR-TT, Chapter I, p. I37

[301] External Drive: EDRP03\Monte Carlo_No Pumping_April 2007\Simulation files and results-Used\

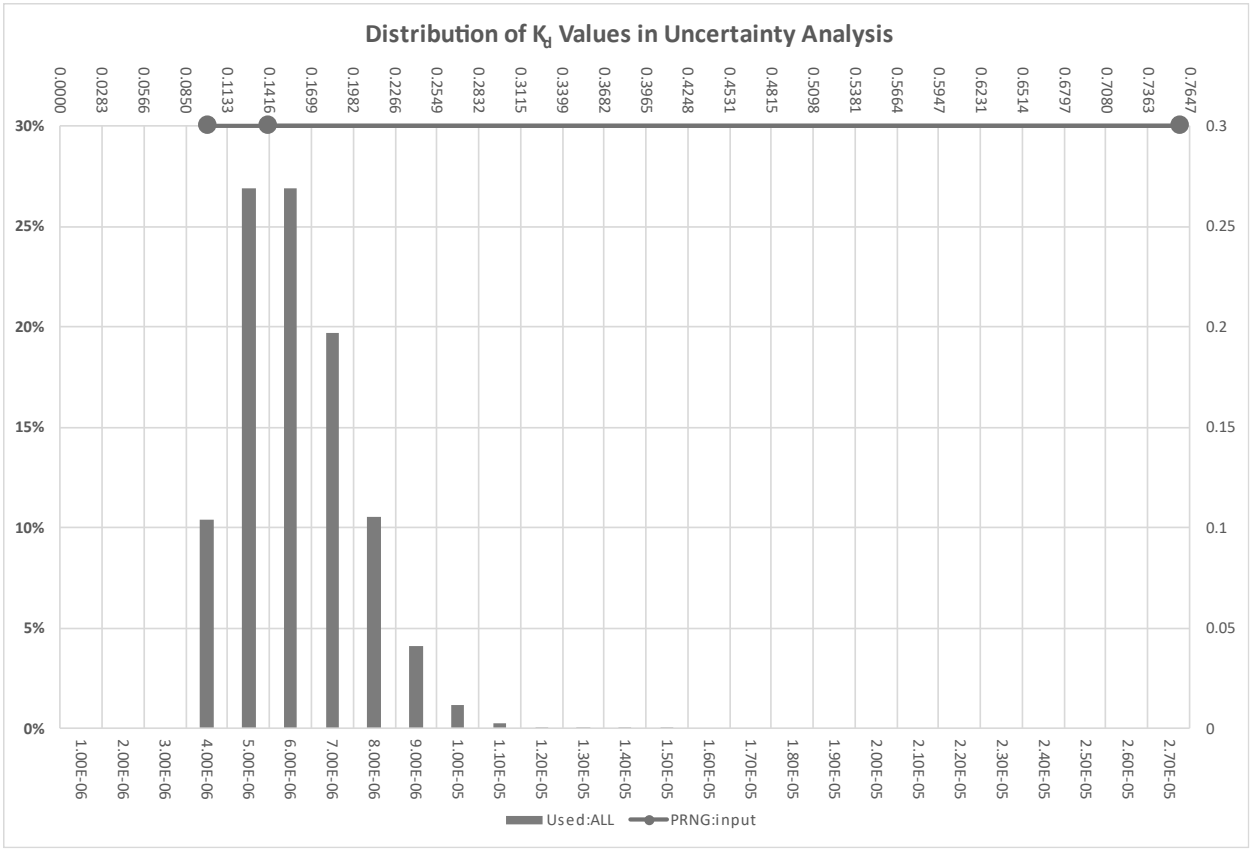

Figure 37: Distribution of K_d Values in ATSDR's Uncertainty Analysis

ATSDR did not provide an explanation for their choice of statistical descriptions for the K_d distribution, which inherently prevented the assessment of a wider range of possible K_d values from the range ATSDR considered reasonable.

Similar conclusions can be drawn for the bulk density and porosity distributions used for generating model inputs for the Monte Carlo realizations.

C-2. Statistical descriptions for the model parameters result in distributions closely resembling the mean values used in the calibrated model

Although there is cell-by-cell variability in the calculated parameter fields for each realization, the contaminant plume migrates based on average conditions along its path. The changes in parameter value from cell to cell would result in varying plume migration patterns from one realization to the next if the mean value of the parameter were to vary between realizations. However, in ATSDR's uncertainty analysis, the mean parameter values for K_d, bulk density, and porosity for all realizations are comparable to the corresponding calibrated-model value, and do not encompass the range of values that ATSDR indicated as reasonable, or even the narrower range that resulted from the probabilistic distributions.

The impact of selecting these statistical descriptions for assigning property values in each model cell is illustrated in Figure 38. This map depicts the K_d values assigned to each model cell in Layer 3, within a focus area that encompasses the model-calculated plume footprint between the source area at well TT-26, for one of ATSDR's uncertainty analysis realizations. Per ATSDR's approach, K_d values are assigned randomly in each cell, sampled from the distribution depicted in Figure 37.

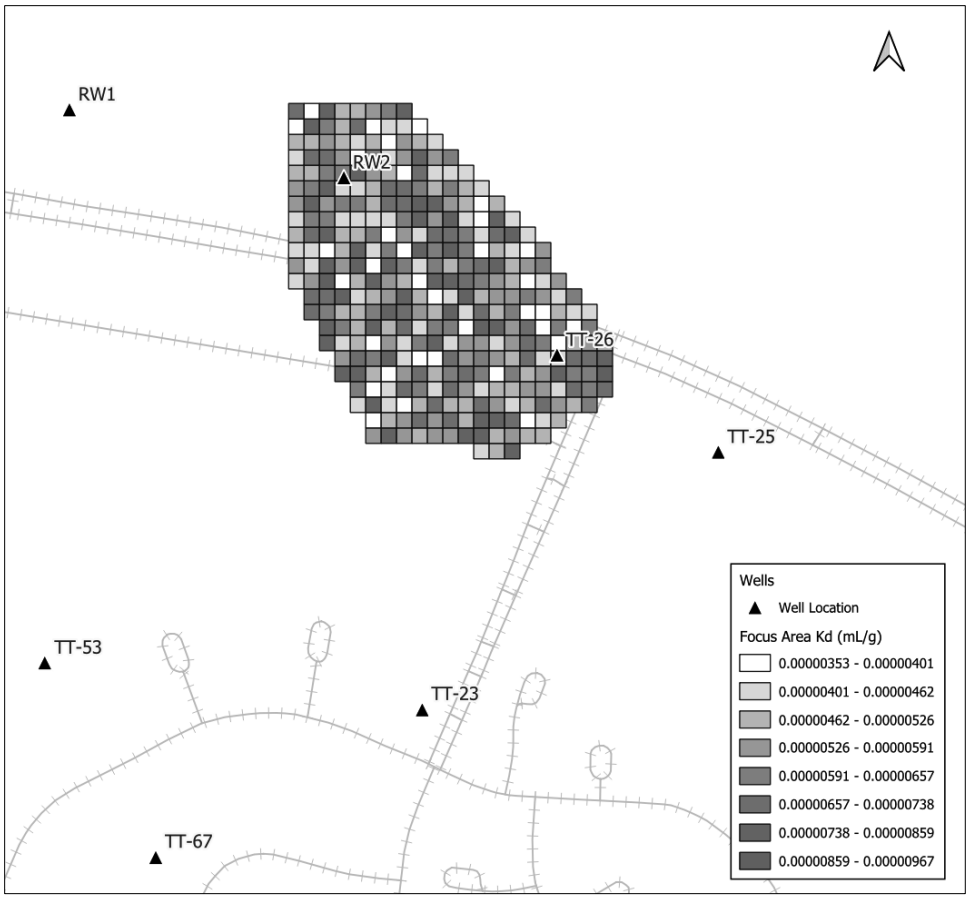

Figure 38: Cell-by-Cell Distribution of K_d Values in Focus Area of ATSDR's Model Layer 3

The mapped spatial distribution of Figure 38 indeed encompasses K_d values within the range indicated in the statistical description of Figure 37. However, when the mean and median K_d values are calculated for the focus area in Layer 3, the corresponding values are 0.159 and 0.154 mL/g, respectively. The mean and median K_d values for the focus area in Layer 1 for all 840 realizations are 0.159 and 0.154 mL/g, respectively.

These values confirm that (a) ATSDR implemented an approach that considered parameter variability within a fraction of the reasonable range of values to be evaluated; (b) this variability resulted in mean K_d values over the entire model and, more importantly for the historical reconstruction, over the critical distance/area between the source and the pumping wells, that did not vary within even the narrow uncertainty range that ATSDR ultimately assessed; and (c) these mean values are very similar to the values used in the calibrated model, which should be expected considering the statistical descriptions ATSDR implemented for these parameters.

I also calculated mean and median values for bulk density and porosity for the cells in the focus area in Layers 1 and 3, indicating the same patterns as K_d. The same conclusions are drawn when values for K_d, bulk density and porosity are calculated for all model layers/cells and all 840 realizations.

Mean and median parameter values for the focus area from 840 realizations based on the defined probability distribution function:

Parameter	Layer	Mean	Median
K_d	1	0.159	0.154
	3	0.159	0.154
Porosity	1	0.2	0.2
	3	0.2	0.2
Bulk Density	1	77,008	77,053
	3	77,008	77,053

Property values varied significantly from cell to cell in ATSDR's analysis, but the overall mean values did not vary. The cell-by-cell variability resulted in a very small variability in plume migration and contaminant arrival times at the water-supply wells, as plume migration patterns and timing are determined primarily by mean property values over the distance/area of interest and not the small-scale variability implemented in this analysis. Hence, ATSDR's uncertainty analysis evaluated only local-scale spatial variability and did not address large-scale variability or, more importantly, mean-value variability that would impact plume migration patterns.

Appendix D: Sampling Data Available During the Historical Period in Tarawa Terrace

Table F13. Simulated and observed tetrachloroethylene (PCE) concentrations at water-supply wells and calibration target range, Tarawa Terrace and vicinity, U.S. Marine Corps Base Camp Lejeune, North Carolina.

[µg/L, microgram per liter; ND, not detected; J, estimated]

Site name	Date	PCE concentration, in µg/L		Calibrated target range, in µg/L
		Observed	Simulated	
[1]RW1	7/12/1991	ND	0.0	0.0–2.0
[1]RW2	7/12/1991	760	1,804	240–2,403
[1]RW3	7/12/1991	ND	0.0	0.0–2.0
[2]TT-23	1/16/1985	132	254	41.7–417
	2/12/1985	37.0	254	11.7–117
	2/19/1985	26.2	253	8.3–82.8
	2/19/1985	ND	253	0.0–10.0
	3/11/1985	14.9	253	4.7–47.1
	3/11/1985	16.0	253	5.2–52.5
	3/12/1985	40.6	253	12.8–128
	3/12/1985	48.0	253	15.4–154
	4/9/1985	ND	265	0.0–2.0
	9/25/1985	4.0	279	0.3–12.6
	7/11/1991	ND	193	0.0–5.0
[2]TT-25	2/5/1985	ND	6.2	0.0–10.0
	4/9/1985	ND	8.6	0.0–2.0
	9/25/1985	0.43J	18.1	0.14–1.4
	10/29/1985	ND	20.4	0.0–10.0
	11/4/1985	ND	20.4	0.0–10.0
	11/13/1985	ND	20.4	0.0–10.0
	12/3/1985	ND	22.8	0.0–10.0
	7/11/1991	23.0	72.6	7.3–72.7
[2]TT-26	1/16/1985	1,580	804	500–5,000
	2/12/1985	3.8	804	1.2–12
	2/19/1985	55.2	798	17.5–175
	2/19/1985	64.0	798	20.2–202
	4/9/1985	630	801	199–1,999
	6/24/1985	1,160	732	367–3,668
	9/25/1985	1,100	788	348–3,478
	7/11/1991	340	670	111–1,107
[2]TT-30	2/6/1985	ND	0.0	0.0–10.0
[2]TT-31	2/6/1985	ND	0.15	0.0–10.0
[2]TT-52	2/6/1985	ND	0.0	0.0–10.0
[2]TT-54	2/6/1985	ND	5.8	0.0–10.0
	7/11/1991	ND	30.4	0.0–5.0
[2]TT-67	2/6/1985	ND	3.9	0.0–10.0

[1]See Figure F6 for location

[2]See Figure F1 for location

Note: Calibration target ranges for analyses listed as not detected are detection limits noted in Table F2

Figure 39: Simulated and Observed PCE Concentrations at Water Supply Wells in Tarawa Terrace (ATSDR, Table F13, Chapter F)

Table F14. Computed and observed tetrachloroethylene (PCE) concentrations in water samples collected at the Tarawa Terrace water treatment plant and calibration target range, U.S. Marine Corps Base Camp Lejeune, North Carolina.

[µg/L, microgram per liter; TTWTP, Tarawa Terrace water treatment plant; ND, not detected]

Date	PCE concentration, in µg/L		Calibration target range, in µg/L
	Computed[1]	Observed	
[2]TTWTP Building TT-38			
5/27/1982	148	180	25–253
7/28/1982	112	[3]104	33–329
7/28/1982	112	[3]76	24–240
7/28/1982	112	[3]82	26–259
2/5/1985	176	[3,4]80	25–253
2/13/1985	3.6	[5]ND	0–10
2/19/1985	3.6	[6]ND	0–2
2/22/1985	3.6	[5]ND	0–10
3/11/1985	8.7	[6]ND	0–2
3/12/1985	8.7	[6,7]6.6	2.1–21
3/12/1985	8.7	[6,8]21.3	6.7–67
4/22/1985	8.1	[5]1	0.3–3.2
4/23/1985	8.1	[5]ND	0–10
4/29/1985	8.1	[5]3.7	1.2–11.7
5/15/1985	4.8	[5]ND	0–10
7/1/1985	5.5	[5]ND	0–10
7/8/1985	5.5	[5]ND	0–10
7/23/1985	5.5	[5]ND	0–10
7/31/1985	5.5	[5]ND	0–10
8/19/1985	6.0	[5]ND	0–10
9/11/1985	6.0	[5]ND	0–10
9/17/1985	6.0	[5]ND	0–10
9/24/1985	6.0	[5]ND	0–10
10/29/1985	6.0	[5]ND	0–10
[2]TTWTP Tank STT-39			
2/11/1985	176	[5]215	0–10

[1] Weighted-average computation

[2] See Plate 1, Chapter A report, for location (Maslia et al. 2007)

[3] Detection limit is unknown

[4] Analysis of tap water sample for Tarawa Terrace, address unknown

[5] Detection limit = 10 µg/L

[6] Detection limit = 2 µg/L

[7] Sample collected downstream of TTWTP reservoir after operating well TT-23 for 24 hours

[8] Sample collected upstream of TTWTP reservoir after operating well TT-23 for 22 hours

Figure 40: Simulated and Observed PCE Concentrations at the Tarawa Terrace WTP (ATSDR, Table F14, Chapter F)

Appendix E: Sampling Data Available During the Historical Period in Hadnot Point and Holcomb Boulevard

Table A4. Water-supply wells with reported detections of tetrachloroethylene (PCE), trichloroethylene (TCE), 1,1-dichloroethylene (1,1-DCE), *trans*-1,2-dichloroethylene (1,2-tDCE), *cis*-1,2-dichloroethylene (1,2-cDCE), total 1,2-dichloroethylene (total 1,2-DCE), or vinyl chloride (VC), Hadnot Point–Holcomb Boulevard study area, U.S. Marine Corps Base Camp Lejeune, North Carolina.[1]

[<, constituent is less than the detection limit. Number following the "<" is the detection limit; —, constituent concentration not determined in laboratory analysis; ND, constituent not detected; J, estimated concentration; D, sample diluted for analysis]

Well name	Sample date	[2]Concentration, in micrograms per liter						
		PCE	TCE	1,1-DCE	1,2-tDCE	1,2-cDCE	Total 1,2-DCE	VC
HP-602	7/6/1984	<1.9	<1.4	<1.3	7.8	—	—	<0.9
	11/30/1984	24	1,600	2.4J	630	—	—	18
	12/10/1984	<500	540	<500	380	—	—	<500
	12/13/1984	3.2	300	—	110	—	—	—
	12/14/1984	<50	340	<50	230	—	—	<50
	2/4/1985	1.5J	38	<10	74	—	—	<10
	11/12/1986	<4.1	2.2	<2.8	14	—	—	<4.9
	1/22/1991	<5.0	0.7J	<5.0	—	—	12	<10
HP-603	12/4/1984	<10	4.6J	<10	<10	—	—	<10
	12/12/1984	<10	<10	<10	<10	—	—	<10
	1/16/1985	<10	<10	<10	<10	—	—	<10
	8/11/1988	<10	<10	<10	<10	—	—	<10
	6/26/1990	<5.0	<5.0	<5.0	—	—	—	<10
	1/22/1991	<5.0	1.0J	<5.0	—	—	<5.0	<10
	9/20/1995	<0.50	3.0	<0.50	<0.50	2.4	—	<0.50
HP-608	12/4/1984	<10	110	<10	5.4J	—	—	<10
	12/10/1984	<10	13	<10	2.4J	—	—	<10
	2/4/1985	<10	9.0	<10	<10	—	—	<10
	11/12/1986	<4.1	66	<2.8	8.5	—	—	<4.9
HP-610	2/4/1985	<10	<10	<10	<10	—	—	<10
	10/1/1992	<1.0	37	—	—	—	—	<2.0
HP-634	12/4/1984	<10	<10	<10	<10	—	—	<10
	12/10/1984	<10	<10	<10	2.3J	—	—	<10
	1/16/1985	10	1,300	<10	700	—	—	6.8
	11/12/1986	<4.1	<1.9	<2.8	2.9	—	—	<4.9
	1/22/1991	<5.0	<5.0	<5.0	—	—	1.0J	<10
HP-637	12/4/1984	<10	<10	<10	<10	—	—	<10
	12/10/1984	<10	<10	<10	<10	—	—	<10
	1/16/1985	<10	<10	<10	<10	—	—	<10
	1/22/1991	<5.0	0.90J	<5.0	—	—	<5.0	<10
	8/26/1992	<5.0	<5.0	<5.0	<5.0	<5.0	—	<5.0
HP-651	1/16/1985	386	3,200	187	3,400	—	—	655
	2/4/1985	307	17,600	<200	8,070	—	—	179
	2/4/1985	400	18,900	<200	7,580	—	—	168
	11/12/1986	45	32	7.0	140	—	—	140
	1/22/1991	53	13	2.0J	—	—	75	70
HP-652	1/16/1985	<10	9.0	<10	<10	—	—	<10
	11/12/1986	<3.0	<3.0	<2.8	<1.6	—	—	<1.0
	1/22/1991	<5.0	<5.0	<5.0	—	—	<5.0	<10
	9/20/1995	<0.50	<0.50	<0.50	<0.50	<0.50	—	<0.50
	12/11/2001	<0.50	<0.50	<0.50	<0.50	<0.50	—	<0.50
HP-653	1/16/1985	<10	5.5	<10	<10	—	—	<10
	11/12/1986	<4.1	2.6	<2.8	<1.6	—	—	<4.9
	1/22/1991	<5.0	<5.0	<5.0	—	—	<5.0	<10
HP-660	12/4/1984	5.0J	210	<10	88	—	—	<10
	12/10/1984	4.4J	230	<10	99	—	—	<10
	1/16/1985	<10	26	<10	88	—	—	<10
	11/12/1986	<4.1	<1.9	<2.8	<1.6	—	—	<4.9
	1/22/1991	<5.0	1.0J	<5.0	—	—	2.0J	<10

[1] See Faye et al. (2010) for a complete tabulation of contaminants in water samples collected at water-supply wells in the Hadnot Point–Holcomb Boulevard study area

[2] Concentrations above the detection limit are highlighted in red

Figure 41: Concentration Data for PCE and its Degradation By-Products in HPHB Water Supply Wells (ATSDR, Table A4, Chapter A)

Table A5. Water-supply wells with reported detections of benzene, toluene, ethylbenzene, or total xylenes, Hadnot Point–Holcomb Boulevard study area, U.S. Marine Corps Base Camp Lejeune, North Carolina.[1]

[<, constituent is less than the detection limit. Number following the "<" is the detection limit; —, constituent concentration not determined in laboratory analysis; ND, constituent not detected; J, estimated concentration]

Well name	Sample date	[2]Concentration, in micrograms per liter			
		Benzene	Toluene	Ethylbenzene	Total xylenes
Hadnot Point Water Treatment Plant Service Area					
HP-602	7/6/1984	380	10	8.0	—
	11/30/1984	120	5.4J	<10	—
	12/10/1984	720	<500	<500	—
	12/13/1984	<1.0	<1.0	<2.0	—
	12/14/1984	230	12J	<50	—
	2/4/1985	<10	<10	<10	—
	11/12/1986	50	<6.0	<7.2	<12
	1/22/1991	17	<5.0	<5.0	<5.0
HP-603	12/4/1984	<10	<10	<10	—
	12/10/1984	<10	<10	<10	—
	1/16/1985	<10	<10	<10	—
	8/11/1988	<10	<10	<10	<10
	6/26/1990	<5.0	<5.0	<5.0	<5.0
	1/22/1991	<5.0	<5.0	<5.0	<5.0
	9/20/1995	<0.50	<0.50	<0.50	<0.50
HP-608	12/4/1984	3.7J	<10	<10	—
	12/10/1984	4.0J	<10	<10	—
	2/4/1985	1.6	<10	<10	—
	11/12/1986	<4.4	<6.0	<7.2	<12
HP-651	1/16/1985	<10	<10	<10	—
	2/7/1985	<10	<10	<10	—
	11/12/1986	<4.4	<6.0	<7.2	<12
	1/22/1991	<5.0	0.9J	<0.5	<0.5
Holcomb Boulevard Water Treatment Plant Service Area					
HP-645	2/4/1985	<10	<10	<10	—
	11/6/1986	20	7.5	ND	ND
	2/17/1987	290	15	38	36
HP-706	9/19/1995	0.60	<0.50	<0.50	<0.50
	1/13/1998	6.1	—	—	—

[1] See Faye et al. (2010) for a complete tabulation of contaminants in water samples collected at water-supply wells in the Hadnot Point–Holcomb Boulevard study area

[2] Concentrations above the detection limit are highlighted in red

Figure 42: Concentration Data for Benzene, Toluene, Ethylbenzene, or Total Xylenes in HPHB Water Supply Wells (ATSDR, Table A5, Chapter A)

Table A18. Selected measured and reconstructed (simulated) concentrations of tetrachloroethylene (PCE), trichloroethylene (TCE), *trans*-1,2-dichloroethylene (1,2-tDCE), vinyl chloride (VC), and benzene at the Hadnot Point water treatment plant, Hadnot Point–Holcomb Boulevard study area, U.S. Marine Corps Base Camp Lejeune, North Carolina.

[µg/L, microgram per liter; J, estimated]

Contaminant	[1]Measured data		[2]Reconstructed (simulated)		[2]Reconstructed (maximum value)	
	Sample date	Concentration, in µg/L	Simulation date	Concentration, in µg/L	Simulation date	Concentration, in µg/L
PCE	5/27/1982[3]	15	May 1982	21	Nov. 1983	39
	7/27/1982[4]	100	July 1982	27		
	12/4/1984[6]	3.9J	Nov. 1984	31		
	2/5/1985[7]	7.5J	Jan. 1985	16		
TCE	5/27/1982[3]	1,400	May 1982	438	Nov. 1983	783
	7/27/1982[5]	19	Aug. 1982	670		
	7/27/1982[6]	21	Aug. 1982	670		
	12/4/1984[5]	46	Nov. 1984	639		
	12/4/1984[6]	200	Nov. 1984	639		
	12/12/1984[6]	2.3J	Dec. 1984	43		
	12/19/1984	1.2	Dec. 1984	43		
	2/5/1985[7]	429	Jan. 1985	324		
1,2-tDCE	12/4/1984[6]	83	Nov. 1984	358	Nov. 1983	435
	12/4/1984[5]	15	Dec. 1984	26		
	12/12/1984[6]	2.3J	Dec. 1984	26		
	2/5/1985[7]	150	Jan. 1985	163		
VC	2/5/1985[7]	2.9J	Jan. 1985	31	Nov. 1983	67
Benzene	11/19/1985[7,8,9]	2,500	Nov. 1985	3	Apr. 1984	12
	12/10/1985[7]	38	Dec. 1985	3		
	12/18/1985[7]	1.0	Dec. 1985	3		

[1]Data from Faye et al. (2010, Tables C11 and C12)

[2]Simulation results represent the last day of each month (e.g., May 31); results reported for simulation month nearest the sample date; refer to Appendix A7 for complete listing of reconstructed finished-water concentrations

[3]Water sample collected at Building NH-1; data reported as unreliable

[4]Water sample collected at Building FC-530

[5]Untreated water

[6]Treated water

[7]Treatment status unknown

[8]Laboratory analysis noted with: "Sample appears to have been contaminated with benzene, toluene, and methyl chloride" (JTC Environmental Consultants 1985)

[9]Data noted with: "Not Representative" (U.S. Marine Corp Base Camp Lejeune Water Document CLW #1356)

Figure 43: Concentration Data for PCE, TCE, 1,2-tDCE, VC, and Benzene at the Hadnot Point WTP (ATSDR, Table A18, Chapter A)

Attachment

Curriculum Vitae

Alexandros Spiliotopoulos, Ph.D.

Senior Associate, Senior Hydrogeologist

Dr. Spiliotopoulos' expertise is quantitative and qualitative analysis of environmental data to support water resources management. He develops and applies analytical and numerical groundwater models, develops novel methods for evaluating water quantity and quality data, and designs and optimizes multiscale remedial systems and monitoring programs. He brings extensive experience in the remediation of nuclear and Superfund sites, providing expert modeling support for RI/FS, RPO, and RD/RA Work Plans; design, performance assessment and optimization of remedial systems for fuel additives, metals, radionuclides and VOCs; as well as water-resource assessment for water-supply development, permitting, and adjudication.

REPRESENTATIVE EXPERIENCE

S.S. Papadopulos & Associates, Inc. *– Rockville, Maryland*

GROUNDWATER REMEDY DESIGN & EVALUATION

U.S. Department of Energy, Hanford, Washington: Technical Lead and Lead Modeler for the River Corridor Operable Units. Designed system alternatives for Remedial Process Optimization, Remedial Investigation/Feasibility Study, and Remedial Design/Remedial Action Work Plans, including the design and implementation of large-scale pump-and-treat systems and/or MNA and other in-situ treatment technologies, to meet short- and long-term goals for river protection and aquifer cleanup. Used in-house enhanced versions of MODFLOW, MODPATH, and MT3DMS to design, evaluate and optimize remedies to meet cleanup objectives. Examples of this work include:

- Developed strategies for implementing EPA's DQO process and statistical evaluations for site closure, considering MNA and/or in-/ex-situ treatment.
- Constructed and calibrated the 100 Areas 3D groundwater flow and reactive contaminant transport groundwater model, using the MODFLOW suite of codes. performed uncertainty analysis using Null Space Monte Carlo to evaluate aquifer restoration timeframe uncertainty.
- Constructed and calibrated groundwater flow and fate & transport models for all Operable Units in the River Corridor.
- Designed sampling and analysis plans and test designs for area-specific hexavalent chromium rebound studies to determine if groundwater remediation activities have met or are on track to meet cleanup goals.
- As part of annual reporting since 2012, conducted (a) site-wide multi-constituent plume delineation by developing/ implementing a systematic approach for data-selection and using in-house transformation-based kriging algorithms; (b) pump-and-treat system performance evaluations including flow and fate & transport modeling, statistical assessment of water-quality data (with covariates for seasonal, river, and pumping effects), monitoring network evaluation and recommendations.
- Contributor to the evaluation of the presence, extent, and mass loading from continuing sources, including analysis of batch tests
- Developed functional criteria and designed critical and optimal networks for monitoring wells equipped with pressure transducers and data loggers to collect continuous water-level data.
- Designed and performed reactive transport simulations using the MT3DMS dual-domain formulation and developing the CTS module for time-varying mass recirculation of extracted contaminants via injection wells.

YEARS OF EXPERIENCE

20+

EDUCATION

» **PhD**, Civil and Environmental Engineering, University of Vermont, 1999

» **BS**, Civil Engineering, University of Patras, Greece, 1994

EXAMPLE AREAS OF EXPERTISE

» Groundwater Remedy Design and Evaluation

» Water Resource Evaluation and Management

» Environmental Data Analysis

» Groundwater Modeling

PROFESSIONAL SOCIETIES

» National Ground Water Association (NGWA)

» American Geophysical Union (AGU)

LANGUAGES

English, Greek

PROFESSIONAL HISTORY

» S.S. Papadopulos & Associates, Inc.: 2004–present

» ADK Consulting Engineers S.A., Hydraulic Engineer: 2000–2004

» University of Vermont, Graduate Researcher in Research Center for Ground Water Remediation Design: 1994–1999

EMAIL

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- For groundwater monitoring under RCRA, developed tools and novel approaches in support of remedial action and contaminant migration pattern evaluations.
- Responsible manager for RCRA progress evaluations in 300 Area, including estimates of concentration trends, yearly mean concentrations, and confidence limits for groundwater wells for the MNA remedy for multiple constituents, and enhanced attenuation (EA) remedy for uranium.

U.S. Environmental Protection Agency (EPA) Region 5: Provides technical support to Region 5 EPA for evaluating groundwater flow, contaminant transport, and remedy performance at multiple Superfund sites. Authored and co-authored reports to support statutory Five-Year Reviews, including recommendations on remedy and monitoring program optimization. Remedial technologies evaluated included pump-and-treat (P&T), monitored natural attenuation (MNA), soil vapor extraction (SVE), air-sparge/biosparge (AS/B).

Sparton Corporation, Albuquerque, New Mexico: Technical Lead and Lead Modeler for performance evaluation and optimization of the remedial design, and assessment of groundwater quality data to evaluate plume migration patterns and effectiveness of remediation of VOCs and metals, including chromium. Constructed and calibrated 3D groundwater flow and contaminant transport for transient site conditions, in support of system performance evaluations. Responsible for all compliance/mitigation efforts and annual reporting to EPA/NMED. Expanded monitoring program to evaluate potential vapor intrusion issues and the presence and migration of 1,4-dioxane within and outside the hydraulic containment zone.

New York Department of Environmental Conservation (NYSDEC), New York: Consulted to the NYSDEC spills department, evaluating and simulating the fate and remediation of fuel spills. Provided hydrogeologic oversight and groundwater flow and fuel-component transport and fate analyses to design and optimize groundwater remedies and the associated monitoring systems to protect sole-source municipal supplies. Projects included:

- **New Hyde Park Site Characterization, Long Island:** Reviewed and supervised site characterization efforts in collaboration with NYSDEC. Developed and calibrated a three-dimensional flow and transport model for a robust pump-and-treat remedy evaluation to support the development of a cost-effective remedial system design to address a MTBE mega-plume.
- **Mineola Site Characterization and Monitoring Plan, Long Island:** Provided technical support for site characterization and design of a monitoring plan for a MTBE plume. Developed and calibrated a three-dimensional flow and transport model for the design of a remedial system to protect a downgradient public supply well. Implemented geostatistical and other methods to evaluate spatial and temporal variations of the magnitude and direction of the hydraulic gradient in support of additional remedy designs to address the MTBE plume.
- **Elmont Site Characterization, Long Island:** Reviewed and supervised site characterization efforts. Designed, coordinated, and performed a rapid mapping methodology for 3D site-characterization and expedited plume delineation. The designed mapping methodology combined direct-push sampling and Quantile-Kriging interpolation techniques. Developed a three-dimensional flow-and-transport model to assess the MTBE plume migration, to evaluate alternative remedial designs, and to monitor natural attenuation.
- **Ronkonkoma Groundwater Flow & Transport Model, Long Island:** Developed and calibrated a groundwater flow and transport model to analyze plume (MTBE, BTEX, TAME, and TBA) migration patterns for at the site, and to monitor the operation of the appropriate Interim Remedial Measure (IRM) system.
- **Uniondale Monitoring Plan, Long Island:** Developed a monitoring plan to delineate the contaminant plume and to assess plume migration characteristics to identify appropriate measures for protection of downgradient receptor wells.
- **Hampton Bays Flow & Transport Model, Long Island:** Applied a three-dimensional flow-and-transport model to assess MTBE-plume migration pathways over time, under varying regional hydraulic gradient conditions. Evaluated the effectiveness of the existing IRM system and recommended system enhancements.
- **West Hempstead Flow & Transport Model:** Developed a transient three-dimensional flow-and-transport model to analyze historical groundwater flow conditions at the site and to identify MTBE plume migration and recovery at the IRM wells. Evaluated system performance, recommended improvements for system operations, and assessed aquifer cleanup times.
- **Gloria Road Groundwater Monitoring Network, Nassau County:** Provided technical support and recommendations for the development of a monitoring network to determine the migration pattern of a MTBE and BTEX plume. Implemented an existing flow model to perform particle-tracking analysis and sensitivity analysis to provide bounding estimates on the lateral extent of the plume pathway and to evaluate whether a proposed monitoring well would intercept the plume.
- **East Patchogue Flow & Transport Model, Suffolk County:** Supervised the development of a three-dimensional flow and transport model to analyze the historical migration of a MTBE plume and to quantify its

relative discharge to a pond near the shoreline and the Long Island Sound. Performed a sensitivity analysis to incorporate variation in pond stage due to tidal effects.

Onondaga Lake Parameter Estimation: Reviewed and expanded parameter estimation efforts for a flow-and-transport model calibration. Evaluated the proposed remedial design and provided recommendations for appropriate design parameters.

Far-Mar-Co Subsite, Hastings Site, Nebraska: Calibration of the groundwater-flow and contaminant-transport model, applying advanced parameter estimation techniques, including regularization, using PEST.

WATER RESOURCE EVALUATIONS

Mississippi vs. Tennessee, City of Memphis and Memphis Light, Gas, and Water Division: Worked on groundwater modeling and ancillary evaluations in support of assessment of water-level declines throughout the aquifer in light of Mississippi's complaint that Tennessee is stealing its groundwater.

North Penn Area 5 Superfund Site near Colmar, in Bucks and Montgomery Counties, Pennsylvania: Compiled and analyzed historical water level data for vertical gradient and particle tracking evaluations in a highly fractured aquifer. Modified the groundwater flow model developed by the USGS, to include additional pumping wells and a former pond area. Developed and assessed various pumping scenarios, via groundwater modeling and particle tracking.

Hanford Natural Resource Trustee Council (HNRTC), Hanford, Washington: Technical lead to conduct a desktop survey to evaluate existing information incorporating a literature study, model outputs, conceptual site models (CSMs), plume maps, and data, to estimate the upwelling of contaminants to the Columbia River at the Hanford Site, in support of a Natural Resource Damage Assessment and Restoration (NRDAR) process undertaken by the Hanford Natural Resource Trustee Council (HNRTC). Developed a hybrid mapping-modeling method for integrating distributed plume information and historical groundwater model results, to provide bounding estimates of contaminant upwelling for the period 1980-2020. The assessment reviewed contaminants of interest (COIs) and produced sitewide plume depictions, hundreds of piecewise-continuous digital maps for the main COIs, mass upwelling graphs illustrating the annual and cumulative mass upwelling for each COI and area, heatmaps illustrating upwelling patterns for each COI and OU, concentration upwelling maps for the main COIs, and maps of annualized temperature distributions for each OU.

Apalachicola-Chattahoochee-Flint River Basin, Florida vs. Georgia Water Dispute: Provided technical support to Georgia's Counsel in reviewing material submitted for consideration in the conflict-resolution litigation at the U.S. Supreme Court. Reviewed model files, numerical and analytical evaluations, and historical data. Provided high-level assessment of current and projected water-resource management practices and plans, and of the impacts to inter-state water allocations.

Confidential Client: Technical Lead for the evaluation of groundwater as a large-scale potential water supply for an international site serving a population of more than 5 million people. Led and conducted hydrogeologic and geochemical evaluations, reviewed geophysical assessments to enhance hydrogeologic interpretations, and developed multiple conceptual models for hypotheses testing by constructing, calibrating and deploying groundwater flow and transport models in a highly fractured aquifer with transboundary considerations. Designed and performed uncertainty analyses using IES for calibrated realizations. Provided recommendations for short- and long-term aquifer testing and monitoring/characterization activities.

U.S. Army Corps of Engineers (USACE), New England District: Technical Lead for the construction and calibration of a MODFLOW-USG groundwater flow model and deployment of mod-PATH3DU to evaluate pumping effects on surface water bodies, assess contaminant migration pathways, areas of influence, and Zone II Wellhead Protection Area for the proposed well. The model will serve as the main tool for evaluating PFAS fate and transport, as part of the RI/FS effort at the site.

Kansas Department of Agriculture Republican River Basin Model, Northwest Kansas: Modified and re-calibrated the existing Republican River Compact Association flow model (focusing on the Northwest Kansas area) to provide an administrative tool for the prediction of impacts to the Republican River from varying future irrigation patterns. Provided technical support to Kansas Department of Agriculture to evaluate future resource allocation and compliance on the basis of a Tri-State water-use agreement.

Kansas Department of Agriculture Solomon River Basin Groundwater Flow Model Evaluation: Supervised and provided recommendations for the development and calibration of two groundwater flow models for the Solomon River basin, to be used as an administrative tool for the management of the available water resources for irrigation purposes.

Montana vs. Wyoming and North Dakota, Tongue River Basin Technical Support: Provided technical support in reviewing groundwater model parameters and results, historical stream flow and outfall data, and other analyses to evaluate aquifer response and stream depletions to irrigation pumping and coal bed methane (CBM) pumping.

Rainelle Power Plant Water Supply Evaluation, West Virginia: As part of the EIS for the construction and operation of this proposed power plant, developed and calibrated a three-dimensional flow model to simulate groundwater flow conditions to evaluate the availability of groundwater as a water source for cooling purposes and to evaluate potential impacts to local pumping wells and river flow. Performed aquifer test analyses to define hydraulic properties that were further refined during the model calibration process. Conducted baseflow analysis to estimate river flow that was attributed to groundwater and to evaluate river water depletions due to pumping. Developed pumping and recharge scenarios to assess the impact of pumping from existing and proposed wells that would supply the power plant, to local production wells and to river flow.

Confidential Client, El Campo, Texas: Performed statistical analyses of isotopes and other chemicals to examine origin of contaminants and plume migration for a site contaminated with TCE, DCE and other chemicals. Constructed, calibrated, and deployed numerical and semi-analytical methods for simulating groundwater flow and contaminant transport (MODFLOW/MT3D and ATRANS), to estimate the contaminant release history at the site, based on recent monitoring data.

ADK Consulting Engineers S.A. – Athens, Greece

Water distribution network, Corfu, Greece: Designed and calibrated a model of the metropolitan water distribution network of the City of Corfu.

Athens Water Supply and Sewerage Company, Greece: Updated the numerical model for the principal mains of the water distribution network of the City of Athens and its suburbs, to simulate water demands associated with the 2004 Olympic Games. The assessment included all recent changes to the network and served as the primary management tool of the water distribution network. Developed a database and a GIS application for data management and interface with the hydraulic simulation model.

Olympic Village, Athens, Greece: Project engineer responsible for the design of the irrigation system of the Olympic Village, including four pumping stations and two storage ponds. Completed the preliminary and final design and tender document preparation for the irrigation system and a treated water storage tank. Evaluated the efficiency of the water distribution network design parameters, and performed complete fire-flow analysis.

Municipality of Lamia, Greece: Proposed an alternative design of the new treated-water storage tank for the City of Lamia for a total capacity of 2,500m³, and developed a GIS application interface for the management of the new treated-water aqueducts.

Pan-Peloponnesian National Stadium of Patras, Greece, Ministry of Culture: Designed the new stormwater management network as part of the stadium reconstruction for the 2004 Olympic Games.

Publications & Presentations

Spiliotopoulos, A., Karanovic, M., Chowdhury, M., Ni, J., and Tonkin, M., 2025, *An Integrated Approach for Developing Contaminant Upwelling Estimates in the Hyporheic Zone*. Presentation at th Waste Management Conference, Phoenix, AZ, March 12, 2025 (in preparation)

Spiliotopoulos, A., Tonkin, M., DiFilippo, E., and Sorel, D., 2025, *Remedy Challenges, Novel Approaches and Lessons Learned from Superfund Waste Sites*. Presentation at th Waste Management Conference, Phoenix, AZ, March 10, 2025 (in preparation)

Muffels, C., S. Panday, C. Andrews, M. Tonkin, and A. Spiliotopoulos, 2022, *Simulating Groundwater Interaction with a Surface Water Network using Connected Linear Networks*. Groundwater, v. 60, no. 6 (November-December), pp. 801-807. doi: 10.1111/gwat.13202

Spiliotopoulos A., E.L. DiFilippo, P. Khambhammettu, D. Hayes, M.J. Tonkin, M. Hartman, K. Ivarson, and J. Hulstrom, 2019. *Web-Assisted Methods and Tools for Efficient Remedy Design and System Performance Evaluation at Hanford*. Presentation at the Waste Management Conference, Phoenix, AZ, March 7, 2019. Received "Superior" paper and "WM2019 Papers of Note Winner" awards. OSTI #23003084

DiFilippo E.L., M.J. Tonkin, A. Spiliotopoulos, W. Huber, and V. Rohay, 2019. *Evaluating Environmental Remediation Performance at Radwaste Sites Using Multiple, Censored Regression Analysis*. Presentation at the Waste Management Conference, Phoenix, AZ, March 7, 2019. IAEA #52043413

Spiliotopoulos, A., R. Shannon, M.J. Tonkin, and L.C. Swanson, 2011. *Evaluation of Temporal Variations in Hydraulic Capture due to Changing Flow Patterns Using Mapping and Modeling Techniques*. Presentation at MODFLOW and More 2011, Colorado School of Mines, Golden, CO. OSTI #1011435

Bedekar, V., M.J. Tonkin, and A. Spiliotopoulos, 2011. *Implementation of a Contaminant Treatment System (CTS) Module in MT3DMS*. Presentation at MODFLOW and More 2011, Colorado School of Mines, Golden, CO.

Khambhammettu, P., M.J. Tonkin, and A. Spiliotopoulos, 2011. *FIELDGEN_D – A Modified 2D Field Generator for Deterministic and Stochastic Groundwater Modeling*. Presentation at MODFLOW and More 2011, Colorado School of Mines, Golden, CO.

Shannon, R., A. Spiliotopoulos, and M.J. Tonkin, 2011. *Estimating Contaminant Migration Pathways Using a Time Sequence of Water Level Maps and Particle Tracking*. Presentation at the 2011 Ground Water Summit and 2011 Ground Water Protection Council Spring Meeting. National Ground Water Association, Baltimore, MD.

Smoot, J.L., F.H. Biebesheimer, J.A. Eluskie, T. Simpkin, M.J. Tonkin, and A. Spiliotopoulos, 2011. *Groundwater Remediation at the 100-HR-3 Operable Unit, Hanford Site, Washington*. Presentation at the Waste Management Conference, Phoenix, AZ. February 27– March 3, 2011. OSTI # 1004613

Spiliotopoulos, A., M.J. Tonkin, D. Shrimpton, J. Blount, T. Simpkin, and J. Hanson, 2010. *Groundwater Modeling in Support of Remedial Process Optimization: Implementing a Developing Conceptual Site Model into Comparative Remedy Analyses*. Presentation at the Waste Management Conference, Phoenix, AZ, March 7-11, 2010.

Spiliotopoulos, A., M. Karanovic, and S.P. Larson, 2008. *Development of Transient Flow Models for the Solomon River Basin*. Presentation at MODFLOW and More 2008, Colorado School of Mines, Golden, CO.

Spiliotopoulos, A., K. Krajenke, N. Hart, J. Haas, D. Cornacchiulo, D. Trego, and M. Tonkin, 2008. *Robust Pump-and-Treat Remedy Evaluation for an MTBE Mega-Plume*. Presentation at the National Ground Water Association (NGWA) Conference on Eastern Regional Ground Water Issues, Ronkonkoma, NY.

Spiliotopoulos, A., K. Krajenke, K. Salafrio, and J. Haas, 2008. *Rapid Mapping to Support Accelerated Site Assessments*. Presentation at NGWA Conference on Eastern Regional Ground Water Issues, Ronkonkoma, NY.

Spiliotopoulos, A., and C.B. Andrews, 2007. *Analysis of Aquifer Test Data – MODFLOW and PEST*. in Groundwater and Wells. (3rd ed.). Sterrett, R.J., ed. New Brighton, MN: Johnson Screens. 812 p. (also presented at MODFLOW and More 2006, Colorado School of Mines, Golden, CO).

Spiliotopoulos, A.A., G.P. Karatzas, and G.F. Pinder. 2004. *A Multi-period Approach to the Solution of Groundwater Management Problems Using an Outer Approximation Method*. European Journal of Operational Research, v. 157, no. 2 (September), pp. 514-525. doi: 10.1016/S0377-2217(03)00239-X

Matsouki, M., G. Germanopoulos, and A. Spiliotopoulos, 2003. *Geographical Information Systems (G.I.S.) Implementation in Water Supply Network Modeling*. Presentation at the XXX IAHR Congress, Thessaloniki, Greece. August 24-29, 2003.

Spiliotopoulos, A.A., G.P. Karatzas, and G.F. Pinder, 2000. *A Biconcave-Decomposition Method for the Optimal Design of Pump-and-Treat Remediation Systems Including the Treatment Plant*. Presentation at the 13th International Conference on Computational Methods in Water Resources, Calgary, Canada. June 2000.

Karatzas, G.P., and A.A. Spiliotopoulos, 1998. *Development of Two Optimization Models: (1) A Multi-period Approach to Solve Engineering Management Problems, and (2) Incorporating Uncertainty in the Decision-Making Process for Optimal Management Design. An Application to a Groundwater Management Problem*. Presentation at the Vermont EPSCoR (Experimental Program to Stimulate Competitive Research) Annual Conference on Science and Technology, University of Vermont, October 6, 1998.